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A REVIEW OF THE BIOLOGICAL FATE AND EFFECTS OF OIL IN COLD MARINE ENVIRONMENTS

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A REVIEW OF THE BIOLOGICAL FATE AND EFFECTS OF OIL IN COLD MARINE ENVIRONMENTS

by

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CHAPTER 1

INTRODUCTION

This report, prepared for Environment Canada (EPS), summarizes current knowledge regarding the biological fate and effects of petroleum hydrocarbons on flora and fauna in cold marine environments. It is intended for use by the Canadian scientific community, including government and industry resource managers and research scientists in both the public and private sectors. Therefore, "cold marine environments" encompass all Canadian arctic and subarctic waters throughout the year and waters bordering Canada's east and west coasts during the winter months. The biological effects of oil spills have been the subject of numerous reviews and several books over the last two decades. The present report does not attempt to add to an already exhaustive literature on this subject, but rather provides a synthesis of existing information in a form suitable for readers seeking an overview of the biological fate and effects of petroleum hydrocarbons. Readers requiring a more in-depth treatment of topics covered in this review are referred to the source literature.

In accordance with the terms of reference for this study, the authors have avoided specific comments on the direction and methods used in current oil spill research, and have not provided recommendations for future research. However, areas where apparent data deficiencies hampered evaluation of the possible significance of biological effects observed during past studies have been identified.

As indicated above, the emphasis in this review has been on the biological fate and effects of oil in cold marine environments. Literature describing the effects of oil spills in freshwater and terrestrial environments has not been summarized, although many of the biological resources discussed in the report occur in these environments at some time in their life history. The scope of the review is further limited to literature relevant to those species found in Canadian marine environments. Brief overviews of the behaviour and physical-chemical fate of oil in marine environments (e.g. spreading and movement of oil slicks and weathering processes) are provided as background information. Readers interested in a more detailed treatment of this subject are referred to Mackay (1984).

This report has been organized according to five distinct habitat types that may be affected by oil spills and well blowouts. Separate chapters deal with:

- Oil on open water
- Oil within the water column
- Oil in benthic environments
- Oil associated with ice
- Oil on shorelines

Within this framework, subsections provide overviews of the behaviour and fate of oil in each habitat, and summarize available information regarding the effects of oil on biological populations and communities found in each habitat. The resources discussed under each habitat were selected on the basis either of (1) their documented primary distribution in marine environments, or (2) the primary focus of the oil effects literature for that organism. For some populations such as fishes and plankton, the most appropriate habitat category for discussion of oil spill effects was obvious. However, other organisms may occur in a range of habitats throughout the year or at different stages in their life history, and could be affected by oil present in several different habitats. For example, some species of arctic seals could be affected by oil on the water surface, within the water column, on shorelines and in areas with ice cover. Seals feeding on benthic prey could also be directly or indirectly affected by oil in this habitat. However, since virtually all of the literature describing the effects of petroleum hydrocarbons on seals has focused on interactions between seals and surface slicks, this resource is discussed in the chapter dealing with oil on open water. The introductions to each chapter describe those situations where populations of certain species may be affected by oil in more than one habitat.

It should emphasized that oil spills are dynamic events. Initially, spills may only affect the water surface, but with spreading, movement and dispersion of surface slicks and progressive weathering of the oil, other habitats and biological communities are eventually affected. These processes may continue well after the actual release of the oil. For example, oil stranded in shoreline habitats and sedimented in shallow benthic environments may re-enter adjacent water column habitats due to reworking of coastal sediments through ice or wave action. The authors therefore recommend that the reader view this organization by habitat categories as simply a convenient approach for presentation of information, and recognize that oil spills are very dynamic events.

This report provides a synthesis of the available information on the sensitivity and vulnerability of each category of biological organism to effects of oil spills, as well as any data on the recovery of populations following exposure. These are the three most important criteria for hazard evaluation. It is important to distinguish between sensitivity and vulnerability. Individuals in a population may be particularly sensitive to effects of petroleum hydrocarbons because very low oil concentrations cause mortality or decrease growth or the ability to reproduce. On the other hand, the population itself may not be particularly vulnerable to oil exposure because individuals are distributed throughout an area much larger than that affected by oil and therefore the numbers of individuals affected would be low. There are other instances where both the sensitivity of individuals and the vulnerability of populations are high. For example, a population of seabirds may be highly sensitive to oil, and at the same time, congregates in a localized area to stage or breed where the risk of oil exposure may also be high.

In addition to discussing sensitivity, vulnerability and recovery for the interactions of each organism with oil, most sections describe the sensitivity of resources in terms of:

- mortality
- behavioural effects
- contamination and physical effects
- physiological effects
- pathological effects
- effects on growth, development and reproduction
- hydrocarbon uptake, metabolism and depuration
- effects of oilspill countermeasures
- indirect effects

During preparation of this report, we relied extensively on other recent reviews of the biological effects of oil spills. Computer and library searches for current literature sources to supplement these reviews were restricted to information published since 1982. Major information sources include: D.C. Malins, ed. (1977); Duval et al. (1981); F.R. Engelhardt, ed. (1985); the proceedings of the bi-annual oil spill conferences sponsored by the American Petroleum Institute, Environmental Protection Agency and U.S. Coast Guard; and the reports of participants in the Baffin Island Oil Spill Program (BIOS). These documents were supplemented by the general scientific literature and information contained in recent Environmental Impact Statements prepared for offshore hydrocarbon development in the Canadian Beaufort Sea, Alaska and east coast of Canada.

CHAPTER 2

OIL ON OPEN WATER

This chapter provides an overview of the behaviour and fate of oil present on the water surface, and summarizes available information regarding the biological fate and effects of oil on the following marine resources: seabirds, seals, sea lions, walruses, whales, dolphins and porpoises. With the exception of non-diving species of birds, these animals may also be exposed to oil present within the water column in either dissolved or dispersed forms. Some marine mammals are also known to feed on benthic prey organisms, and diving ducks can consume benthic fauna in shallow waters. However, most of the literature describing the effects of oil on these resources has focused on interactions with surface oil slicks, and this is the main reason for reviewing relevant information for these species in the present chapter.

It is also emphasized that some of the flora and fauna discussed in other chapters may be affected by surface oil slicks at some time during their life cycle. The most notable examples are: (1) the eggs of some species of fish that float on the water surface (Section 3.4); (2) kelps that are anchored to the sea floor, extend through the water column and have fronds floating on the surface (Section 4.2); (3) marine-associated mammals (e.g. polar bears, sea otters) that may be affected by surface oil (Section 6.4); and (4) shoreline and wading birds that may contact surface oil slicks during some activities (Section 6.5).

SECTION 2.1

AN OVERVIEW OF PHYSICAL AND CHEMICAL PROCESSES IN OPEN WATER

Sy Ross

The biological effects of oil spills in open water are determined by the properties of the spilled oil, the type, size and location of the spill, and prevailing meteorological conditions. All of these factors also affect the behaviour and fate of the oil on the water surface and in the water column (see Section 3.1). This chapter provides an overview of oil spill dynamics and focuses on those factors that have a major influence on the impact of spills on birds, marine mammals and other organisms.

2.1.1 The Important Processes

The major phenomena that influence the impact of spills on biota at the water surface are spreading, movement, and persistence of oil. Oil spreading increases the size of the affected area, and thereby increases the probability that greater numbers of individuals and proportions of populations will contact the oil. Oil movement, which is the progressive displacement of the surface slick(s) by wind and current forces, increases the risk to biota which otherwise may be unaffected by a spill. The persistence of oil on the water surface determines the quantity of oil that is present at any time to interact with biological resources.

Other physical processes which "weather" the oil, such as evaporation, dissolution and emulsification, also affect the impacts of spills. Weathering alters certain properties of the oil and causes progressive changes in both its chemical toxicity and physical consistency. However, weathering processes are much more important for their influence on spreading and persistence than for their effect on toxicity, because the biological effects of surface oil are primarily related to its physical rather than chemical properties.

In any discussion of the behaviour and fate of oil spills, it is important to emphasize the fundamental axiom that "no two spills are alike". This is particularly true with respect to the two different types of major oil releases that can occur in offshore environments. The first is a continuous discharge of oil (e.g. oil well blowout) where a comparatively small but steady flow of oil to the water surface continues for a relatively long period of time. The second is a batch spill (e.g. tanker accident) in which a large quantity of oil reaches the water surface over a relatively short period. The behaviour of oil reaching the water surface varies substantially between these two events. Because batch spills occur far more frequently than continuous releases, only the former are discussed in this chapter. In addition, continuous discharges tend to be less persistent than large batch spills primarily because the initial thickness of oil slicks near blowouts is very small, and thin slicks are known to disperse rapidly in open sea conditions.

2.1.2 Oil Spreading

The most notable feature of any marine oil spill is the surface spreading phenomenon. Numerous models of oil spreading behaviour and its dependence on oil properties and environmental conditions have been developed over the last two decades (e.g. Blokker 1964; Fay 1969; Mackay et al. 1980). All of these models relate the properties of the oil (density, viscosity and interfacial tension) to its spreading on calm water. The more comprehensive models also include an oceanic diffusion term to describe spreading behaviour in more realistic sea conditions.

The model developed by Mackay et al. (1980) includes the assumption that as the slick spreads, it forms thick patches (about 1 mm) that contain 90 percent of the oil volume in 10 percent of the area. These patches are surrounded by thin sheens (about 1-100 μm) containing 10 percent of the oil volume in 90 percent of the area. The feature is consistent with observed spill behaviour. The model can be used to predict temporal changes in the radius and area of surface slicks resulting from different size spills. As evident in Figure 2.1-1, oil initially spreads rapidly and eventually reaches a maximum slick size (the slick area depicted in the figure includes both the thick and thin areas described above). Both the area covered by a slick and the time required for it to reach maximum size (equilibrium) increase with oil volume. However, the relationships shown in the figure do not apply to oils that will not spread over the water surface. This category includes oils that have a pour point (i.e. the temperature below which the oil does not flow) which exceeds the ambient temperature. The pour point of an oil can be naturally high, or it may be increased by evaporative losses of some hydrocarbons during weathering.

2.1.3 Oil Movement

In addition to the natural tendency of most oils to spread over a water surface, surface slicks may be transported away from the site of a spill by water currents. These currents usually combine residual movement and wind-induced surface movement. Mackay (1984) reports that the rate of oil slick movement may be increased by a factor of 3.6 ± 0.5 percent of the wind speed measured 10 m above the water surface. However, the final vector of oil movement depends on the combined influences of residual currents and wind-induced surface currents. Over relatively long distances, Coriolis forces (due to the earth's rotation) cause the direction of wind-induced slick motion to deviate some 10-15° to the right of the wind direction (in the Northern Hemisphere). This type of oil movement occurs both during and after the period of oil spreading. In nearshore marine waters, the movement of oil slicks is also affected by tidal currents, river outflows and longshore currents.

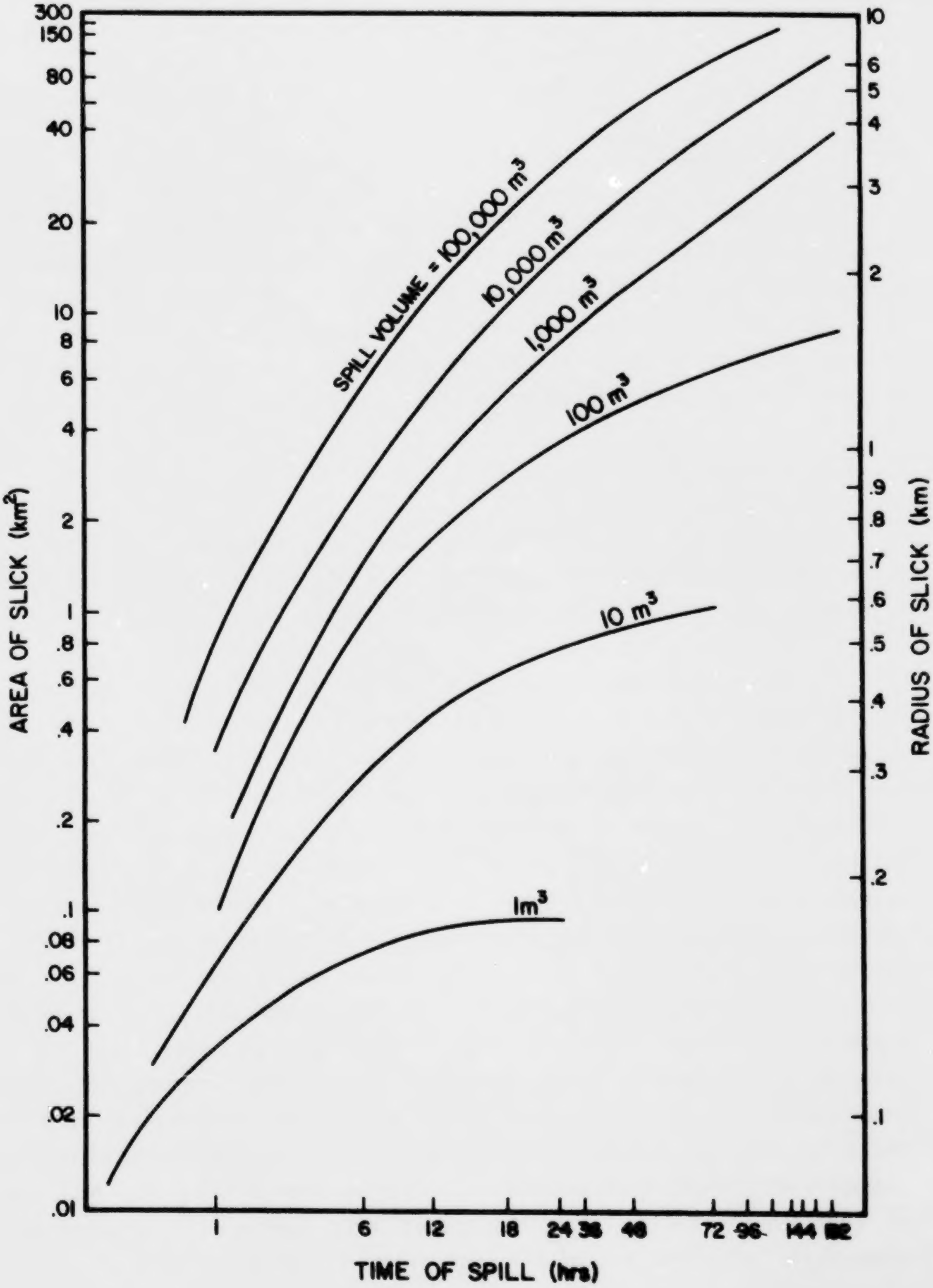

Figure 2.1-1 Slick Dimensions Versus Time After Spill For Spills of Typical Oils of Various Volumes (After Mackay et al. 1980)

2.1.4 Oil Persistence

The persistence of oil on the water surface is controlled primarily by three processes: (1) evaporation; (2) natural dispersion of oil into the water column; and (3) the formation of water-in-oil emulsions. The first two processes remove oil from the water surface, but compete with the latter process which tends to stabilize the oil slick and reduce losses from the surface. Each of these processes is briefly summarized below.

2.1.4.1 Evaporation

Evaporation is one of the most important processes that affects the properties and therefore the behaviour of spilled oil. It is also the most intensively studied and predictable process (Mackay 1984).

The evaporation rate of an oil slick is controlled or affected by: (1) the temperature of the oil and the air; (2) the surface area of the oil in contact with air; (3) the thickness of the oil; (4) wind speeds, and perhaps most importantly; (5) the concentration and vapour pressure of the individual components of the oil. Although there have been many studies of oil evaporation rates, they have all followed a similar approach of determining an overall "mass transfer coefficient" as a function of environmental conditions. The most useful studies have been conducted by Mackay at the University of Toronto (e.g. Nadeau and Mackay 1978; Stiver and Mackay 1983). In these studies, the volume or mass fraction of oil evaporated are related to an exposure coefficient (combining time, oil volume and area, and the mass transfer coefficient to the atmosphere) and to the pressure-concentration behaviour of the oil. The unique aspect of this approach is that it permits the results from a variety of laboratory evaporation experiments to be easily extrapolated to actual environmental conditions with a relatively high degree of confidence. Table 2.1-1 illustrates this approach in predicting the evaporative loss from a 1-mm slick of crude oil as a function of sea state (Exxon 1983).

TABLE 2.1-1

EVAPORATIVE OIL LOSS OF LIGHT AND MEDIUM
CRUDE OIL SLICKS AS A FUNCTION OF SEA STATE
(after Nadeau and Mackay 1978)

SEA STATE	OIL LOSS (Percent)					
	Exposure Time = 6 h			Exposure Time = 24 h		
	5 °C	15 °C	25 °C	5 °C	15 °C	25 °C
LOW	16	21	28	23	32	38
MEDIUM	23	32	39	28	37	44
HIGH	26	35	42	29	38	45

Assumptions: Slick Thickness = Volume/Area = 1×10^{-3}m
Oil Density = 8356 km $(m^3)^{-1}$

(Source: Exxon 1983)

2.1.4.2 Emulsification

When most crude oils are spilled at sea, they tend to form water-in-oil emulsions (occasionally referred to as "chocolate mousse" or simply "mousse"). Emulsification occurs in the presence of mixing energy such as that provided by wave action. During emulsification, seawater is incorporated into the oil in the form of microscopic droplets. This water intake results in several undesirable changes to the oil. First, there is a significant increase in the bulk volume of the oil (usually up to a 5-fold increase), greatly increasing the amount of oily material that can contaminate shorelines and biological resources. Secondly, there is a marked increase in fluid viscosity (up to 100-fold). The much higher viscosities greatly inhibit the natural dispersion of oil and make the application of chemical dispersants useless.

The mechanisms and rates of oil emulsification are poorly understood. Through some mechanism, the mixing energy associated with waves causes small water droplets to become entrapped in the oil layer. It is likely (but not certain) that this process occurs in a manner similar to the natural dispersion of oil droplets into the water column, as both are mixing phenomena that require wave action. The two processes may be competitive. Emulsification tends to inhibit dispersion because the former increases oil viscosity, while the latter is very negatively dependent on viscosity.

2.1.4.3 Natural Dispersion

The dispersion of oil into the water column by natural forces is the most important process controlling the long-term fate of oil slicks at sea. In conjunction with evaporation, this process reduces the volume of oil on the water surface, thereby influencing the potential extent of surface and shoreline contamination.

Although natural dispersion is a poorly understood process, it is known that oil/water interfacial tension, oil viscosity, oil buoyancy and slick thickness all interact to affect the ability of a particular oil to disperse naturally. Sea state is also an important factor controlling the rate and amount of dispersion. Even light, non-viscous oils do not rapidly disperse under calm conditions. On the other hand, even the heaviest, emulsified oils can disperse over a period of time in heavy seas with frequent breaking waves.

The predicted range of dispersion rates expected for a "typical" crude oil as a function of sea state, slick thickness, and degrees of evaporation and emulsification (i.e. oil "age") is illustrated in Figure 2.1-2. These relationships were derived by S.L. Ross Environmental Research Limited (1984) using the results of several researchers including Milgram et al. (1978), Buist (1979) and Audunson (1980). The "typical" crude oil used for the analysis had a fresh oil density of 850 kg $(m^3)^{-1}$, a viscosity of 11 mPas at 15 °C and

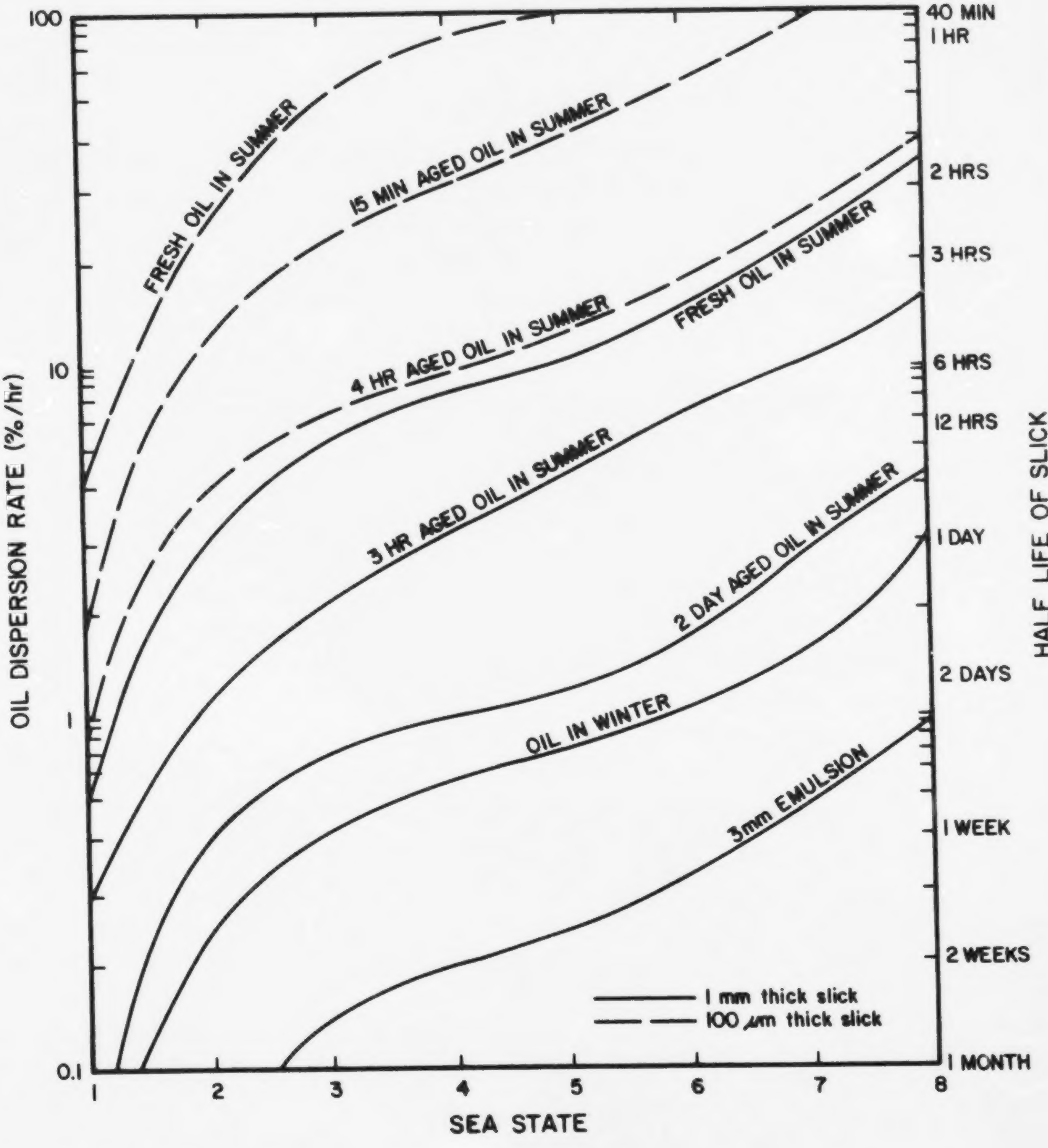

Figure 2.1-2 Predicted Natural Dispersion Rates For Typical Crude Oils (Source: Ross 1984)

175 mPas at 0 °C. The implications of the competitive nature of the emulsification and natural dispersion processes are demonstrated in this figure. A 1-mm slick of fresh, unemulsified oil in summer would have a half-life of only 6 h (i.e. 50 percent will disperse in 6 h under average summer sea states), while the same slick when partially emulsified (3-mm emulsion) would have a half-life of 2 weeks at similar sea states.

2.1.5 Effects of Cold Temperatures on Oil Fate and Behaviour

The primary effect of cold temperatures is to increase viscosity. On the basis of available information, this would decrease spreading rates, greatly reduce evaporation and natural dispersion rates, and probably increase rates of emulsification. The net effect of these changes would be to increase the persistence of oil on the water surface, thereby increasing the risk of interactions with biological resources in this habitat.

SECTION 2.2

MARINE BIRDS

Ken Trudel

Aside from being most visibly affected by oil spills, marine birds are the most vulnerable of marine organisms to their effects. This situation is even more pronounced in cold environments, where both the sensitivity and vulnerability of marine birds to oil are expected to be higher than in temperate regions due to the combined effects of low temperatures and the presence of ice.

The degree of impact of any given spill on populations of marine birds is a function of several factors including: (1) the sensitivity of the members of the population to oil contamination; (2) the vulnerability of individuals and of the population as a whole to oil; and (3) the potential of the population to recover from the effects of the spill. The following sections summarize the state of knowledge regarding the sensitivity, vulnerability, and recovery potential of marine birds in cold environments.

2.2.1 Sensitivity of Birds to Oiling

Birds are exposed to oil via contamination of the plumage and ingestion. Oil contaminates plumage when individuals come into contact with slicks or oil in shoreline habitats (see also Section 6.6). Contaminated birds may then in turn transfer some of the oil to eggs or nestlings. Oiled birds attempt to clean their plumage by preening and in so doing, may ingest some of the oil. Birds whose plumage has not been contaminated may ingest oil or hydrocarbons by consuming contaminated prey. The effects of exposure to oil via these vectors as well as the bioaccumulation kinetics of ingested oil are discussed below. Birds exhibit a number of responses to contact with oil, ranging in severity from minor disruption in osmoregulation to potentially life-threatening loss of the insulating property of plumage. Although a wide variety of responses of birds to oil have been observed in field and lab studies, emphasis here is placed on effects such as mortality or reduced reproductive success that have clear population or ecological consequences.

2.2.1.1 Surface Contamination of Birds

The single most important vector through which oil affects seabirds is contamination of plumage. Birds may become oiled either through direct contact with oil slicks in offshore or nearshore waters, or through contact with oil stranded in coastal environments where they nest or feed. The manner by which surface oil affects birds was probably first elucidated by Portier (1934) who recognized that oiled ducks rapidly became hypothermic when forced to swim in 10 °C water. The plumage of an oil-contaminated bird becomes matted and loses its waterproofing properties, so that the bird becomes less buoyant. At the same time, the plumage loses some of its insulating properties. Although oil contamination may be sufficient to cause birds on water to sink and drown, loss of insulation is the primary cause of death, particularly in cold climates. Birds, like mammals, are homeotherms and maintain a constant core temperature within fairly narrow limits. These core temperatures are well above ambient temperatures in cold marine environments. For example, certain seabirds and waterfowl have body temperatures ranging from 40-43 °C (King and Farner 1961). The heat required to maintain these temperatures is produced by metabolic activity of tissues, while heat loss is minimized by the insulating properties of the plumage. The loss of insulation as a result of oiling leads to rapid heat loss. Portier (1934) observed that this heat loss in an oiled duck was both rapid and severe, and reported a decline in body temperature from 42 °C to 22 °C within 30 min when a bird was exposed to 10 °C water.

If the core temperature drops below the critical lethal temperature (21-23 °C in chickens and other fowl; Sturkie 1961), death from hypothermia ensues quickly. Lightly oiled birds can initially compensate for this heat loss by increasing their rate of metabolic heat production. However, unless a bird can offset this increased energy expenditure by increasing its feeding rate, the increased metabolic rate is supported at the cost of catabolism of the body's lipid reserves. When the lipid reserves are exhausted, protein metabolism begins and unless the insulating properties of the plumage are restored, death from exhaustion, starvation, and hypothermia follows rapidly. If oil can be removed from the plumage and its insulating properties partly restored, a reduction in metabolic demands can be achieved (Hartung 1967). This process appears to require 5-10 days, although Hartung (1963) reports that only a portion of the oil is removed and metabolic rate does not return to its pre-contamination level. Clearly, lightly oiled birds may be able to clean themselves before their energy reserves are exhausted and hence may survive, while more heavily oiled birds will be unable to remove sufficient oil from their plumage and will succumb to hypothermia.

The simple presence of oil on the plumage is not by itself sufficient to cause death. Smith and Bleakney (1968) observed lightly oiled purple sandpipers (Erolia maritima) on a Nova Scotia shore, but did not document any mortality of contaminated individuals. Similarly, pink-footed geese contaminated with fuel oil were reported by the Smithsonian Institute (1972), but there was no evidence of mortality. The likelihood of mortality appears to be strongly determined by both the degree of plumage contamination and the ambient air and water temperatures.

The risk of mortality of oiled birds is clearly greater on water than in air. Portier (1934) reported that a heavily oiled duck remained in good health for many days at an air temperature of 13 °C, but rapidly became hypothermic when forced to swim in water at 10 °C. There is an often quoted sentence in Tuck (1961) that states "a small amount of oil on a bird's feathers can bring about mortality". However, at present there is little quantitative information regarding the relationship between the amount of oil on a bird's plumage and the likelihood of mortality. Some circumstantial information is available from Hartung (1963) who found that the level of contamination of the plumage of dead oiled ducks ranged from 0.6 to 0.7 g oil kg^{-1}, suggesting that even this small amount of oil might cause death. However, it is not known if these birds died from exposure to oil or from some other cause indirectly related to oil exposure. Moreover, some of the oil may have accumulated in the plumage after death.

There is ample evidence that in nearshore habitats, oiled birds will remain on land even if this isolates them from their food source (Hartung 1967). On shore, oiled birds appear to be able to survive indefinitely at air temperatures from 10 to 25 °C. Even at -26 °C, oiled birds can survive for extended periods (Hartung 1967). A theoretical relationship between the level of oiling of a bird's plumage, air temperature and the likelihood of mortality is shown in Figure 2.2-1. This figure indicates that birds succumb quickly to acute hypothermia at high levels of oiling and low temperatures. At somewhat higher temperatures and lower levels of contamination, birds can compensate metabolically for increased heat loss, but will eventually die unless they can clean themselves and restore the insulating properties of their plumage. Lightly oiled individuals will not be adversely affected in terms of survival at warmer air temperatures.

2.2.1.2 Surface Contamination of Eggs

The hatchability of the eggs is reduced when an oiled adult bird transfers some oil from its plumage to the egg surface. The toxic effect of oil on embryos has been known since the 1930s, but it was not until the 1950s that Rittinghaus (1956) reported that indirect oiling of offspring might cause mortality during oil spills. Birkhead et al. (1973) observed that an oiled great black-backed gull (Larus marinus) was able to clean itself, but that the eggs it was incubating failed to hatch, thereby demonstrating that oiling at levels too low to cause mortality of adults may be sufficient to cause mortality of embryos.

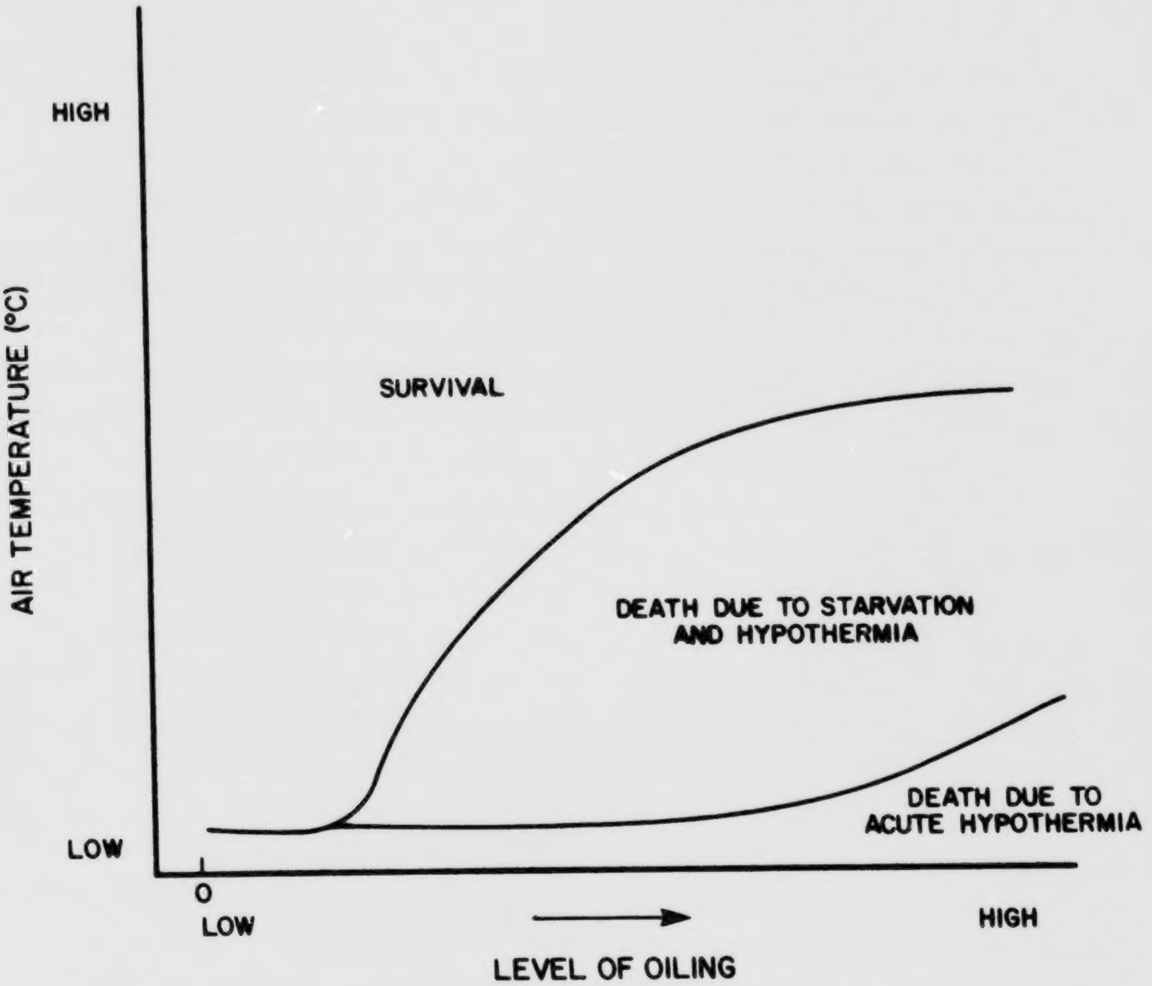

Figure 2.2-1 Theoretical Relationship Between Air Temperature, Level of Oiling, and Survival of Marine Birds

The toxic effects of oil on embryos have been demonstrated in a wide variety of species including the mallard and common eider duck, the laughing gull and great black-backed gull, and the Louisiana heron. Although some interspecific variations in sensitivity exist, all of the above species are highly sensitive to oil (Stickel and Dieter 1979). It was earlier believed that surface oiling of eggs interfered with the oxygen supply to the embryo and caused death by suffocation. However, Hoffman (1978) demonstrated clearly that crude and refined oils exert a chemotoxic effect, possibly mediated by aromatic species of hydrocarbons.

Eggs are sensitive to extremely small amounts of oil. Hartung (1965) was the first researcher to quantitatively examine the effects of oiling on eggs, and found that a surface coating of as little as 10 mg of medicinal mineral oil caused mortality in mallard embryos. Albers and Szaro (1976) demonstrated that as little as 1 µL of No. 2 fuel oil caused death in some mallard eggs, while 50 µL killed all the eggs. There appears to be some variability in toxicity among different oils, but for many crude oils and refined petroleum products, a dose of as little as 50 µL oil per egg is sufficient to kill a large proportion of the population. It should be emphasized that some crudes and refined products retain their embryo-toxicity even after several weeks of weathering. Szaro et al. (1979) showed that as little as 50 µL of No. 2 fuel oil or Prudhoe Bay crude caused 80-100 percent mortality in bird eggs after these oils had weathered for a period of four weeks.

Since birds are probably capable of surviving with several grams of oil on their plumage (especially when they are on shore), it is likely that a lightly oiled breeding bird could contaminate its egg(s) with enough oil to kill the embryos. For this reason alone, reproductive failure is highly probable if adults are oiled during the breeding season. This situation is exacerbated in the Arctic by the fact that ducks incubate unhatched eggs for longer than normal periods and, therefore, are unlikely to renest during the short breeding season if nests or eggs are contaminated with oil.

2.2.1.3 Ingested Oil

Birds ingest hydrocarbons by preening their contaminated plumage and by ingesting oil-contaminated food. The toxic effects of ingested oil have been implicated in the deaths of birds during actual spills, and various laboratory studies have also shown that ingested oil can lead to physiological and histopathological effects, reproductive impairment and death.

(a) Mortality

By feeding oil to healthy birds, several workers have been able to duplicate the pathological conditions observed during autopsies of dead birds found at spill sites. However, it is usually difficult to determine whether the effects of oiled plumage or the effects of ingested oil are the primary cause of mortality. There is evidence that a combination of effects may cause death in some situations.

The specific cause of death following ingestion of oil is not well understood. However, a variety of pathological conditions have been associated with oil ingestion including:

1) Gastrointestinal conditions such as hemorrhage, enteritis, ulceration, and necrosis of the intestinal mucosa;
2) Hepatic degeneration such as fatty degeneration of the liver;
3) Osmoregulatory disruption such as dehydration and nephritis;
4) Lung diseases such as pneumonia; and
5) Adrenal dysfunction.

Certain of the pathological conditions listed above have been attributed to the actual toxic effects of the ingested hydrocarbons, while others have been attributed to stress. Schiber (1981) attempted to rationalize the combined effects of petroleum toxicity and stress as shown in Figure 2.2-2, suggesting that pathological changes to the intestines, liver, and adrenal

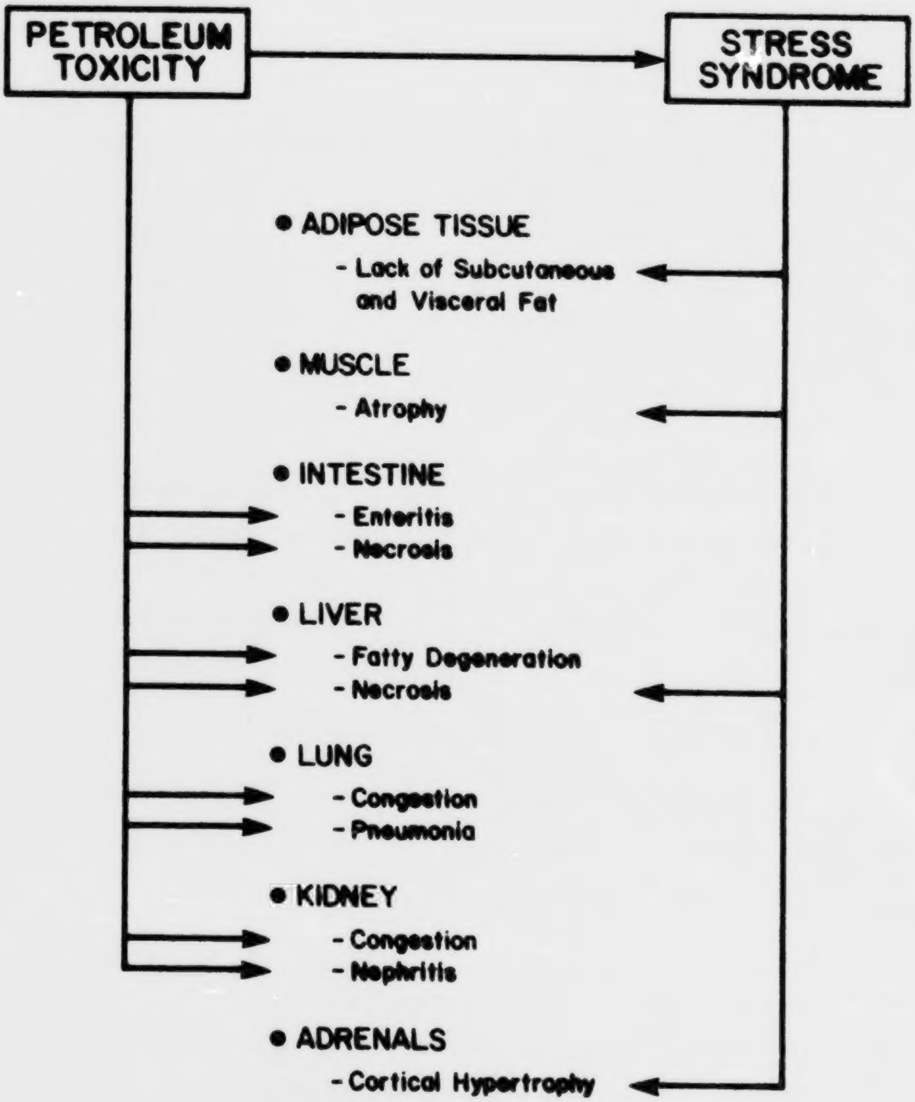

Figure 2.2-2 Causal Factors in Various Lesions Observed in Oil-Exposed Marine Birds at Autopsy (Redrawn from Schiber 1981)

cortex could be manifestations of stress rather than a direct effect of petroleum hydrocarbons.

Few studies have examined the lethal toxicity associated with acute or chronic ingestion of petroleum hydrocarbons. The best information to date regarding the lethal dose of petroleum is from Holmes; Cronshaw; and Crocker (unpublished data, in Holmes and Cronshaw 1977), who found that ducklings died after consuming food contaminated with 5-20 percent South Lousiana crude oil (which corresponds to an exposure rate of 5-20 g oil ingested $kg^{-1}day^{-1}$) after 2-5 days of exposure. Hartung (1967) observed some mortality in Peking ducks after much smaller doses (1-2 mL refined oil products kg^{-1}), although this mortality may have been due to the effects of refining additives rather than the effects of naturally-occurring petroleum hydrocarbons.

Birds examined after marine oil spills often show symptoms of acute dehydration. In freshwater environments, waterfowl can acquire sufficient water to maintain osmotic balance through passive absorption of water in the gut. Osmotic balance is apparently maintained by excretion of water and electrolytes by the kidney in the form of hyposmotic or isosmotic urine. However, birds have a limited capacity to produce hyperosmotic urine and, therefore, in environments where birds are exposed to hyperosmotic drinking water (such as the ocean or haline or alkaline prairie lakes or sloughs), birds osmoregulate by actively taking up Na^+ and water from the gut and excreting excess Na^+ and other electrolytes via the nasal glands. Several authors (e.g. Miller et al. 1978; Peakall et al. 1980) have demonstrated that ingestion of petroleum hydrocarbons results in failure of osmoregulation, as indicated by elevated plasma Na^+ levels. This breakdown is apparently due to the effect of petroleum hydrocarbons on active Na^+ transport systems in the intestinal mucosa and nasal gland. The former is responsible for the uptake of Na^+ and water, while excess electrolytes are excreted from the latter. The results of several studies demonstrate that as little as 0.2 mL (0.5 mL kg^{-1}) of certain crude oils is sufficient to interfere significantly with pump activities of both the intestine and nasal glands, but there is no evidence that the level of inhibition is dose-related at higher doses. It appears that at least some marine species compensate for the reduced specific activity of nasal gland tissue with hypertrophy of the gland (Miller et al. 1978; Peakall et al. 1980) so that total gland activity is depressed only slightly and for a short time. In these species, osmoregulatory failure is transient, with elevated plasma levels returning to background levels within several days (Miller et al. 1978; Peakall et al. 1980). Hypertrophy of the nasal gland and recovery of osmotic balance takes place over a period of 3-5 days.

Most of these studies used ingested doses of 0.5 mL per individual, which are lower than those quantities that could be ingested by a moderately oiled bird while preening. It is not clear from the above studies whether the degree of the responses or their duration are affected by the amount of ingested oil. It is evident, however, that the threshold dose for osmoregulatory effects is sufficiently low that these effects would occur frequently in birds contaminated with oil in marine environments.

(b) Reproduction

There appears to be little information to suggest that egg production in marine birds is affected by ingestion of oil under actual spill conditions, but there is clinical evidence indicating that reproductive disruption occasionally occurs. As mentioned above, birds ingest oil while preening oil-contaminated plumage or when feeding on oil-contaminated food. Oiled birds can preen oil from their feathers in 5-10 days and, therefore, may only be exposed to oil for a short period. On the other hand, if a population's food supply is affected, individuals may ingest contaminated food over a period of several weeks or months. Ingestion of oil causes a number of responses that may influence overall reproductive success. These include: (1) delayed oviposition; (2) reduced rates of egg production; (3) reduced hatching success; and (4) the production of eggs with thin shells.

The effects of short-term hydrocarbon ingestion, as might be experienced by a contaminated bird that has ingested oil during preening, have been investigated in studies where birds are given a single dose of oil. Several authors have demonstrated that egg laying ceases or is delayed for several days in individuals receiving a pulse dose of crude oil or a refined petroleum product (e.g. Hartung 1965; Ainsley et al. 1981). Cessation of egg laying is only temporary and resumes again after a short period. Interference with egg laying may be caused by as little as 0.2 mL of crude oil or refined product per bird. The degree and duration of this effect is dose-dependent. Grau et al. (1977) found that 200 mg of Bunker C oil per bird caused a 3-day reduction in laying in Japanese Quail, while a dose of 500 mg per bird affected egg laying for two weeks or longer. Similarly, Hartung (1965) reported that oviposition in mallards and domestic ducks was arrested for two weeks following ingestion of 2 mL kg^{-1} of lubricating oil. It should be noted that small doses of oil may not affect the overall number of eggs produced. Ainsley et al. (1981) showed that dosages large enough to cause delays in egg laying still did not significantly reduce the number of eggs laid by the treated birds.

Hatching success is also reduced following ingestion of oil. Grau et al. (1977) observed that a single dose of 0.2 g Bunker C oil per bird caused a severe reduction in hatching rate that lasted for three days. Similarly, Ainsley et al. (1981) reported a small but significant reduction in hatching success in Cassin's auklets treated with 0.6 g of Bunker C oil per bird. Oil doses as low as 0.1 g per bird also reduced hatching success, although these effects only lasted a few days.

Eggshell thinning results from oil ingestion and can contribute to reduced breeding success. As little as 400 mg of Bunker C oil per bird may cause a significant reduction in eggshell thickness. However, it is not clear if the degree or duration of this effect is related to the quantity or type of ingested oil.

Studies conducted to date show that even a small amount of oil ingested by a lightly oiled bird during preening activities is sufficient to affect reproduction. For small doses, these effects are of short duration and might only result in an overall reduction in breeding success if exposure coincides with the onset of breeding. Available data also suggest a dose-dependent effect of oil on reproduction. Oil doses in the range of 1-10 g per bird could be lead to reproductive failure in an arctic breeding bird. It is also expected that chronic ingestion of petroleum associated with the ingestion of contaminated food could cause reproductive effects at daily doses much lower than the 0.1-1.0 g oil per bird that caused measurable effects during pulse exposure experiments. Unfortunately, there is very little information on the effects of chronic ingestion of hydrocarbons on the reproduction of seabirds. In one study, Harvey et al. (1982) demonstrated that when domestic ducks were fed a diet containing 5 g oil per 100 g of food (roughly equivalent to an ingested dose of 5 g oil per bird per day) for more than 40 weeks, there were severe reductions in the rate of egg production and eggshell thickness. These observations are not surprising since much smaller pulse doses of oil have been shown to cause similar disruptions in reproduction.

2.2.1.4. Hydrocarbon Bioaccumulation Kinetics

The potential for bioaccumulation of ingested petroleum hydrocarbons in birds has been directly or indirectly examined in studies of the assimilation of specific hydrocarbons from the diet and the metabolic degradation and excretion of assimilated hydrocarbons. Several authors have conducted studies of the accumulation of hydrocarbons in birds (Lawler 1978; McEwan and Whitehead 1980; Tarshis and Rattner 1982). Although the experimental design of these studies was inappropriate to assess assimilation efficiency or retention of specific hydrocarbons, some information regarding the metabolic degradation of hydrocarbons has resulted from each investigation.

Gorsline et al. (1981) examined hepatic degradation of naphthalene in mallards that had been exposed to chronic doses of up to 5 percent South Louisiana crude oil in their diet for 50 days. These authors reported that total hepatic metabolic activity increased with increasing dosage, and at the maximum exposure rate of 6.1 mL of oil $kg^{-1}d^{-1}$, the maximum hepatic metabolic capacity was 0.1 g naphthalene degraded $kg^{-1}d^{-1}$. This research indicated that there were no significant differences between the specific activities of hepatic tissue at the different dosage rates, and that the differences in total hepatic activity were due to dose-dependent hepatic enlargement. Figure 2.2-3 shows total hepatic napthalene metabolic activity as a function of daily ingested dose for mallards. Metabolizing activity of the liver increases in a non-linear manner with ingested dose. Whole body metabolism of naphthalene is not a first-order process even over this narrow range of dose rates, and appears to become somewhat saturated at higher dose rates. This conclusion is, however, based on extrapolation of *in vitro* studies and requires confirmation by *in vivo* investigations.

2.2.1.5 Spill Countermeasures

A range of countermeasures are available to minimize the impacts of oil spills on bird populations. Chemicals can be used to disperse the oil into the water column before it reaches birds or their habitat, or to reduce the persistence of oil at or near important bird habitat. Bird-scare devices may be used to temporarily disperse birds away from oil-contaminated areas. Small numbers of oil-contaminated birds can be cleaned and released. The effects of chemical dispersants and various rehabilitation techniques on bird populations are discussed in the following sections.

(a) Chemical Dispersants

As with oil, birds may be affected by dispersants if these chemicals contaminate their plumage or eggs, or if birds ingest dispersants alone or as a dispersant/oil (D/O) mixture. At present, information on dispersant toxicity is sparse, although some qualitative information is available.

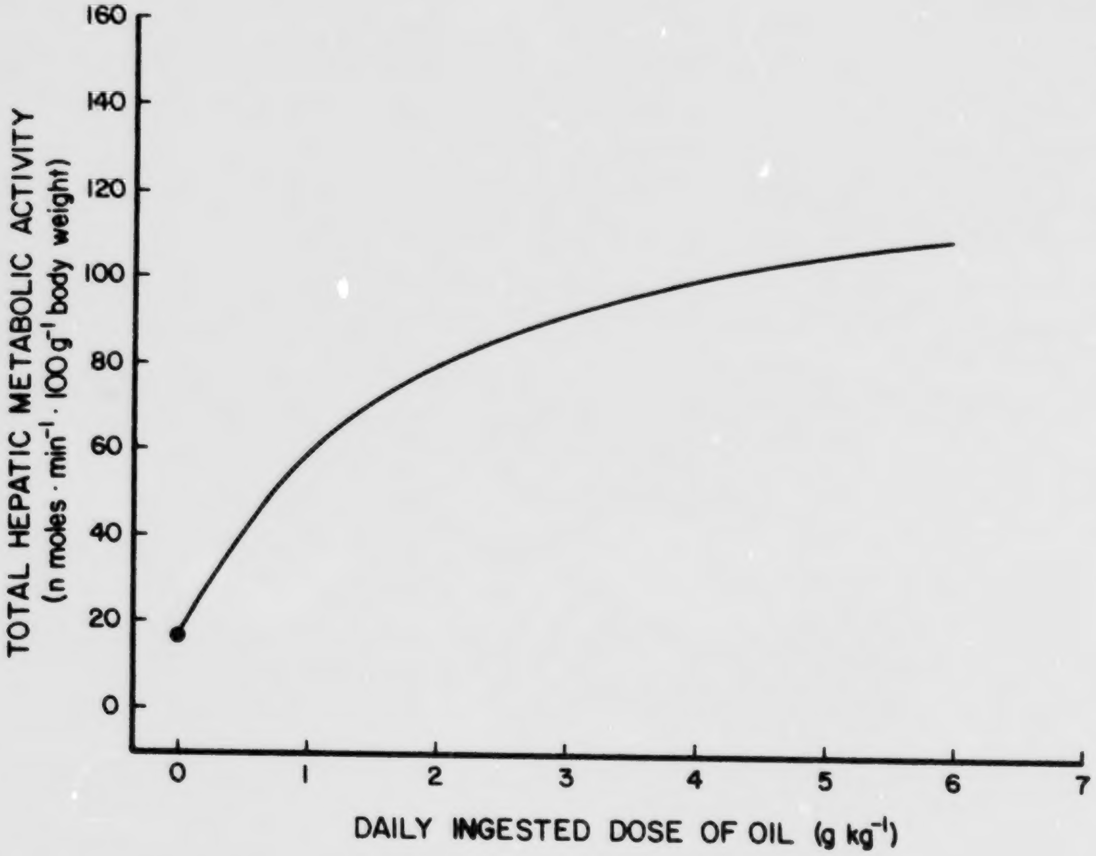

Figure 2.2-3 Total Hepatic Metabolic Activity for Naphthalene Degradation as a Function of Daily Dose in Mallards Fed Chronically on South Louisiana Crude Oil (From Gorsline et al. 1981)

In a study of the effects on birds of externally applied oil and D/O mixtures, Lambert et al. (1982) found that in air, D/O mixtures caused no greater increase in metabolic rate than did oil alone. On the other hand, Miller et al. (1978a and b) found that gulls showed significantly greater weight loss when treated with D/O mixtures than with oil alone, suggesting that D/O mixtures might be somewhat more effective than oil alone in causing a breakdown in the insulative properties of plumage. Lambert et al. (1982) demonstrated a severe and rapid breakdown in waterproofing when dispersant alone was applied to a bird's plumage, although no increase in metabolic rate was observed when the treated birds were out of the water. This indicated that dispersants by themselves did not seriously affect the insulating capacity of the plumage of birds in air. Apparently, contamination of the plumage with dispersant only caused a loss of water-repellent properties of the surface plumage, while the underlying plumage remained unaffected. There was no indication of whether the loss of waterproofing in surface plumage could be reversed by preening or of the time required for this to occur, nor was there any indication of whether this effect was dose-dependent because the plumage retained its flexibility and did not become matted when it was still wet, it could still insulate the bird from cold air.

Although little research has examined the lethality of ingested dispersants or D/O mixtures on birds, studies by Butler et al. (1979) and Peakall et al. (1981) have shown that D/O mixtures had no greater effect on growth, organ weight, corticosterone levels or plasma thyroxine levels in gulls than did oil alone. Other authors have used different indices, but inappropriate experimental design prevent interpretation of the results of these investigations. Several recent studies have shown that dispersant alone had little effect on hatchability when applied to the surface of eggs, and that embryo-toxicity of D/O mixtures was no greater than that of the oil itself (Albers 1979; Albers and Gary 1982).

The database regarding the effects of dispersants and D/O mixtures on birds is very limited. Moreover, the effects of this countermeasure will vary with the type of dispersant used. However, some trends are apparent from studies conducted to date. First, the effects of D/O mixtures are no greater than those of oil by itself, irrespective of whether the oil is acting via surface contamination of eggs or adults or through ingestion. Secondly, there is also some evidence that direct contamination of plumage by dispersants in both nearshore and offshore environments could have relatively serious effects on birds.

(b) Rehabilitation

Early efforts at cleaning oil-contaminated birds were generally unsuccessful due to: (1) inappropriate cleaning techniques; (2) lack of organization among the participants; and (3) attempts to rehabilitate both healthy and severely contaminated birds. Techniques have been improved over the years, and it has been recognized that bird rehabilitation is an interdisciplinary exercise involving knowledge of cleaning techniques, veterinary medicine, and animal husbandry. Environmental groups tend to be optimistic regarding the prospects for mounting successful oiled-bird rehabilitation campaigns in the future, although the scientific community is less optimistic about the success of such efforts (Edmondson 1981).

Although it is clear that some oiled birds can be cleaned and released, the likelihood of successful cleaning remains low. In most cases, the majority of oiled birds recovered along a shoreline already suffer such severe dissipation that they are unlikely to survive if cleaned. This situation is even more severe in colder climates where dissipation will proceed very quickly due to rapid heat loss. Among healthy or lightly oiled birds, rehabilitation of ducks and geese has been highly successful, but the success rate with seabirds (e.g. alcids) is low. Finally, successful bird rehabilitation requires significant organization and resources. In a major spill situation where thousands of birds can be oiled, the level of resources required to clean contaminated birds would far exceed those currently available, particularly in remote areas. Since there is adequate knowledge and resources to rehabilitate some birds, the most practical rehabilitation strategy may be to focus efforts on rare or endangered species. However, rehabilitation is not a practical option for the protection of large numbers of birds (Clark 1984).

2.2.2 Vulnerability of Individuals and Populations to Oiling

For any given spill, certain species of birds will be more vulnerable than others. The vulnerability of a population is a function of two sets of variables (1) the habits of individual birds that determine the likelihood of contact with oil (e.g. amount of time on water, dive rather than take flight when disturbed), and (2) the degree of congregation of individuals in the population(s) that determines whether a large or small proportion of the birds may be contaminated with oil.

Historical data from past spills clearly demonstrate that auks and seaducks are more vulnerable to oiling than are gulls or terns (Vermeer and Vermeer 1975; Duval et al. 1981). Auks and certain other seabirds spend most of the year at sea, and come ashore only in certain seasons to breed. On the other hand, shorebirds and some waterfowl spend a significant proportion of time in terrestrial or backshore habitats. When at sea, auks spend much of their time on the water surface, while gulls and terns spend a large proportion of their time in the air and are less likely to contact oil (Bourne 1968). Auks and seaducks dive to feed and when alarmed, and this habit makes these species more vulnerable to oiling than birds that feed on the wing and take flight when disturbed. Marine birds vary widely in their vulnerability to oil contamination. Auks and seaducks appear to be most vulnerable, while other species such as gulls, gannets, cormorants and certain waterfowl are less vulnerable. Those species of marine birds that spend a significant proportion of their time on shorelines are also vulnerable to oil that may strand in these habitats.

Populations of auks are particularly vulnerable to oil because they are highly gregarious and spend much of the year congregated in certain locations. Colonial breeders such as murres or certain auklets congregate in huge breeding colonies that may contain hundreds of thousands to millions of birds. They may also congregate in offshore foraging areas, both during the breeding season and while overwintering. Certain waterfowl are more widely dispersed during the breeding season, but they congregate at other times of the year. For example, common eiders are present in very large numbers in Canadian coastal waters during winter. An extreme case occurs with populations of greater and lesser snow geese on the marshes of the lower St. Lawrence River near Cape Tormente, Quebec. A major proportion of the North American populations of both species congregate and stage within an area of a few tens of square kilometres for several months in the fall. Even a minor spill at this location and time could have a major impact on these populations. On the other hand, some species do not congregate. Certain species of ducks such as the mallard or teal are widely scattered throughout the year, and it is unlikely that a significant proportion of their populations would be affected by even a very large spill.

In the Arctic, the presence of sea ice contributes to the vulnerability of certain species of birds. Lead systems and polynyas are important in the migration and overwintering of some species (Barry 1970; Vermeer and Anweiler 1975; Stirling 1980) because these openings in the ice cover permit access to food resources. In fact, the development of these patches of open water probably determines the timing of northward migration of certain species. Reliance on such leads causes these populations to become congregated, and hence if such openings in the ice cover become contaminated with oil when large numbers of migrant birds are present, the effects on the populations could be severe.

2.2.3 Recovery Potential

In contrast to marine organisms such as plankton and fish, the time required for recovery of marine bird populations following oil spills is extremely long. Recovery of zooplankton or phytoplankton populations is rapid due to (1) immigration of individuals from adjacent unaffected areas and (2) rapid rates of reproduction. On the other hand, a breeding colony of seabirds is unlikely to restore itself by recruiting large numbers of individuals from neighbouring colonies because nest site tenacity* is highly developed in many species. Certain populations have a number of superfluous adults that can be recruited into the breeding populations in the event of sizeable losses (Dunnet 1982), but in the absence of these superfluous adults, population recovery must be solely through reproduction.

The reproductive capacity of seabirds is low compared to other bird species. Some birds such as auks, although long-lived and having a high survivorship at all ages, produce only a single egg each year. On the other hand, seaducks have a somewhat higher reproductive capacity because they produce several offspring per year. The recovery rate of the latter group is therefore somewhat faster than that of other seabirds (Leppakoski 1973), although it is still slow in comparison with groups such as finfish which produce thousands of eggs per adult each year.

Nettleship (1981) recently suggested that colonial nesting may be partly an adaptation to the need to transfer information concerning the location of food resources in a patchy environment. If this is true, a severe reduction in the size of a bird colony may adversely affect the survival of the remaining members by decreasing the ability of the colony to obtain food.

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*the tendency for a bird to return to the same nest year after year.

SECTION 2.3

SEALS, SEA LIONS AND WALRUSES

Lois Harwood

This section summarizes available information concerning the effects of petroleum hydrocarbons on seals, sea lions and walruses (pinnipeds). The topic is presented within the chapter describing the fate and effects of oil in open water, because much of the literature discussing effects of oil on pinnipeds is based on events that involved their exposure to surface oil. However, pinnipeds could be affected by oil in each of the five habitat types discussed in this review. The marine mammals discussed here spend much of their time in the water column, and also in shoreline or ice-covered areas during the mating, pupping and moulting periods. Some species also forage extensively in benthic environments.

There are three families of pinnipeds. Their representatives in Canadian waters are listed below:

PHOCIDAE (earless or hair seals)

Ringed seal	Phoca hispida
Bearded seal	Erignathus barbatus
Grey seal	Halichoerus grypus
Harbour seal	Phoca vitulina
Harp seal	Phoca groenlandica
Hooded seal	Cystophora cristata
Northern elephant seal	Mirounga angustirostris
Ribbon seal	Histriophoca fasciata
Spotted seal	Phoca largha

OTARIDAE (sea lions and fur seals)	
Northern sea lion	*Eumetopias jubata*
Northern fur seal	*Callorhinus ursinus*
ODOBENIDAE (walruses)	
Walrus	*Odobenus marinus*

2.3.1 Sensitivity of Pinnipeds to Oil

More is known on the effects of exposure of pinnipeds to oil than about any other marine mammal. The majority of this information has been recently summarized and reviewed by Duval et al. (1981), ESL (1982a), Smiley (1982) and Richardson et al. (1983), and includes both laboratory and *in situ* investigations and studies conducted in response to actual oil spill events. The susceptibility of pinnipeds to oil spills varies with species, with the time of the life cycle when exposure occurs, and with the circumstances surrounding the event.

2.3.1.1 Mortality

Mortality of pinnipeds has occasionally been reported following marine oil spills, but data unequivocally linking deaths with the presence of oil has seldom been available. For example, seal and sea lion mortality was only documented following two of 44 crude oil spills reviewed by Duval et al. (1981). Seal pups were killed as a result of the CHRISTOS BITAS spill off Wales (Bourne 1979b), and one dead seal was reported after the ANTONIO GRAMSCI incident in Latria (Anon. 1979a,b). In several other cases, the link between oiling of seals and mortality was not established (Warner 1969; Anon. 1970; Duguy and Babin 1976; Geraci and St. Aubin 1980; Parsons et al. 1980; Duerden and Swiss 1981; Reither 1981; Smiley 1981). Mortality of northern elephant seals, California sea lions, and northern fur seals was reported following the 1969 Santa Barbara crude oil blowout, but critical reviewers were unable to attribute this mortality directly to the presence of oil (Brownell and Le Boeuf 1971; Le Boeuf 1971; Simpson and Gilmartin 1970). Monitoring programs initiated a year after the event indicated no significant increase in mortality of oiled seals, and demonstrated that oiled sea lions gave birth to normal pups (Holmes 1969; Straughan 1970).

Seal mortality following exposure to oil in laboratory studies was documented by Smith and Geraci (1975). Three ringed seals were exposed to a 1-cm thick layer of Norman Wells crude oil in a laboratory holding pen, and all individuals died within 71 minutes. However, the test animals showed signs of stress associated with captivity, and it was believed that this may have intensified the effects of oil exposure (Geraci and Smith 1976). This and other investigations also suggest that seal mortality may be greater if individuals contact oil during periods of natural stress such as poor feeding conditions, heavy ice conditions, moulting, or when individuals are extensively parasitized (Kenyon 1975; Geraci and Smith 1976; Williams 1978).

2.3.1.2 Behaviour

It is not known if pinnipeds are able to detect oil, although the numerous observations of contaminated seals (see Geraci and St. Aubin 1980; Smiley 1982) suggest they do not avoid oil, or cannot avoid it because of the requirement to haulout on shore or ice. There are no available data on recognition, attraction or avoidance of oil at sea by pinnipeds (Engelhardt 1985). While it has been demonstrated that in many instances seals do not avoid spilled oil, Burns and Frost (1979) provide circumstantial evidence that some bearded seals, ringed seals and walruses may occasionally avoid fuel-contaminated waters.

Oil could affect the ability of females to recognize pups, thereby resulting in abandonment and starvation of young (Cowles et al. 1981). However, grey seals have been observed to continue nursing oiled pups (Davies 1949; Davis and Anderson 1976). Although there is little information regarding the effects of oil-contaminated food on feeding behaviour of pinnipeds, Geraci and Smith (1976a) noted that ringed seals continued to accept fish throughout a period when capsules of oil were concealed in their prey. Geraci and Smith (1976a,b) noted an increase in the breathing rate, forced exhalations at the surface, back arching, irritability and increased aggression in ringed seals exposed to Norman Wells crude oil in the immersion study described earlier.

2.3.1.3 Physical Effects

The physical effects of contamination of seals by oil have been described following laboratory investigations and in the oil spill case history literature. These effects include coating of pelage and impairment of mobility, and eye and nostril irritations. However, the occurrence and extent of these effects vary among species and with the circumstances surrounding oil exposure. Viscous oil is more likely to affect locomotion and breathing, and is more probable in arctic waters because low ambient temperatures increase oil viscosity (Engelhardt 1985). This effect may be offset, however, by the fact that arctic crude oils tend to be of low to medium weight.

Davies (1949) reported that mortality of two grey seal pups contaminated by a thick layer of fuel oil was likely due to reduced buoyancy and impeded mobility. Warner (1969) also reported that harp seals heavily coated with Bunker C in the Gulf of St. Lawrence had difficulty swimming and probably died of exhaustion. It was reported that elephant seal pups encrusted with crude oil and debris after the Santa Barbara blowout had difficulty opening their eyes (Anon. 1969). In addition, about 24 harp seals may have died of suffocation when nose and mouth openings were plugged with Bunker C after the grounding of the ARROW in Chedabucto Bay, N.S. (Anon. 1970).

However, oil is gradually lost from pelage and body orifices when seals relocate to uncontaminated areas. For example, 40 percent of the elephant seals tagged (n=63) following the Santa Barbara blowout were resighted alive within 1 to 15 months of the event, and oil was observed on only one individual (Le Boeuf 1971). Survival of the oiled seals was also comparable to that of an unoiled control group. Similarly, Davis and Anderson (1976) reported that grey seal pups covered with weathered oil did not shed their moulting pelage until the contaminated fur layer had abraded. However, the seals were virtually free of oil after the mat of oil and fur was shed.

During oil ingestion experiments described by Geraci and Smith (1976), harp seal pups with fur that had been stained yellow when the animals rolled in oil-contaminated snow, returned to nearly normal white colouration within 24 h after being moved to an area with clean snow.

Ringed seals exposed to a 1-cm slick of fresh Norman Wells crude in an Arctic field holding pen for 24 h showed no evidence of mechanical damage during exposure, and their coats showed virtually no visible evidence of oil immersion after 3 or 4 days in a clean-water holding pen (Geraci and Smith 1976). This may have been because Norman Wells crude is relatively light, highly volatile and has a low viscosity, and is therefore less likely than Bunker C to cause extensive physical fouling in an unweathered state. However, the six seals that were exposed to the crude oil during the above investigation showed severe eye irritation. Seven or eight minutes after the addition of the crude oil to the experimental pen, one seal began to lacrimate and blink excessively, and eye irritation became apparent in the other seals shortly thereafter. Within 4 h, all seals were lacrimating and squinting, and all seals had severe conjunctivitis and swollen nictating membranes at the end of the 24-h oil exposure. In addition, four seals showed evidence of corneal erosions or ulcers. However, within 3 h after the seals were placed in clean seawater, most squinting and lacrimation had subsided, and their eyes showed no signs of irritation by 20 h. Geraci and Smith (1976) suggested that at least some of the eye damage appeared to have been caused by volatile components of the Norman Wells crude, and that continued exposure to oil may have resulted in more severe and possibly permanent eye disorders.

2.3.1.4 Effects on Growth and Reproduction

Many of the physiological effects described in the next section may have implications for growth and reproduction in pinnipeds. However, as indicated earlier, survival of oiled seal pups was comparable to a control group following the Santa Barbara blowout (Le Boeuf 1971), and grey seals continued to nurse oiled pups (Davies 1949; Davis and Anderson 1976). While numerous aborted California sea lion foetuses were found at San Miguel Island following the Santa Barbara blowout, these losses could not be linked to the event or the presence of oil (Simpson and Gilmartin 1970). California sea lion pup mortality noted at San Miguel Island by Brownell and Le Boeuf (1971) was within the range of natural mortality during the year of the spill.

2.3.1.5 Physiological Effects

Oil contamination of the pelage of pinnipeds could alter the thermoregulatory regime through an increase in heat loss to the surrounding medium. Pinnipeds that rely primarily on fur for thermal insulation (fur seals) are most susceptible to thermoregulatory stress as a result of oil contact (Geraci and St. Aubin 1980). These species have a waterproof fur coat that holds a layer of air against the skin for insulation. Oil contamination can result in matting, clumping and derangement of the fur, and a subsequent loss of waterproofing and buoyancy. This may lead to chilling, hypothermia, exhaustion and death (Kenyon 1975; Kooyman et al. 1977; Williams 1978). The duration of time that an oiled fur seal can maintain its core temperature depends on several factors such as extent of contamination, degree of exposure to cold water and air, life history stage and the general condition of the affected individual.

Experimental oiling of subadult fur seal pelts doubled the thermal conductance (Kooyman et al. 1977). Decreases in the insulative capacity of the fur caused by oil contamination leads to a subsequent increase in metabolic rate. For example, light oiling of 30-40 percent of the coat of fur seals caused a 50 percent increase in basal heat production (Kooyman et al. 1977). Increased metabolic rates following oil exposures are stressful and impose additional food consumption requirements or utilization of energy stores at greater than normal rates.

On the other hand, earless seals and sea lions depend on blubber for insulation (Frisch et al. 1974) and the insulative contribution of the pelage is relatively small. However, this varies with species, age, general health and season (Irving and Hart 1957; Frisch et al. 1974; Geraci and Smith 1976). Oiling of ringed seal, bearded seal and California sea lion pelts did not alter their thermal conductance (Oritsland 1975; Kooyman et al. 1977). Body temperature of ringed seals immersed in Norman Wells crude oil for 24 h, and 3- to 4-wk-old harp seal pups coated with crude oil for seven days, remained stable and within the normal range (Geraci and Smith 1976). Subsequent *in vitro* examination of the pelts indicated that thermal conductance had not changed as a result of oil contamination.

Walruses are probably least susceptible to thermo-regulatory stress because they rely exclusively on thick skin and blubber for insulation. However, it is not known if oil contact affects other thermoregulatory mechanisms in these mammals such as peripheral blood circulation.

Although light oiling of adult seals and sea lions probably does not significantly alter the underwater thermal conductance of their pelts, some species may be very susceptible to oil contamination during certain periods of their life history. For example, some newborn phocids (e.g. harp and ringed seals) rely on lanugo (pre-natal fur) for protection against wind chill until a sufficient blubber layer develops and homeothermy stabilizes during the first weeks of life (Oritsland and Ronald 1973). Oiling of pups at this age could result in loss of insulative capacity, chilling and hypothermia (Smiley 1982). Geraci and Smith (1976) found that core temperatures in 2- to 4-wk-old whitecoat harp seals with a 2.5 to 5.0 cm layer of blubber were not affected by a brushed-on coating of crude oil. However, the effects of this treatment may have been different in newborn seal pups up to one week of age, since they have little blubber (Engelhardt and Ferguson 1980) and rely on their pre-natal fur and metabolic activity for thermal balance (Blix et al. 1979; Oritsland and Ronald 1973).

Fur also acts as a wind barrier for adult seals, and may be particularly important during the annual spring moult and obligatory haulout of ringed seals (Geraci and Smith 1976; Oritsland 1975). However, Oritsland (1975) demonstrated that Norman Wells crude oil did not disrupt the insulative capacity of ringed seal pelts at varying wind speeds. Since any coating of dark material increases the radiation transmittance of light-coloured fur (Oritsland 1975), changes in solar heating of the skin of hauled-out seals may occur, although their potential significance has not been documented.

2.3.1.6 Pathological Effects

While laboratory exposures of ringed and harp seals to oil have shown that petroleum hydrocarbons are absorbed and distributed throughout tissues of test animals (Section 2.3.1.7), no major pathological effects resulted from either immersion in oil or oil ingestion (Geraci and Smith 1976). Histopathological investigations showed some evidence of damage of kidney tissue in one seal, but liver damage was minimal and no intestinal lesions were reported. The high hydrocarbon concentrations in the urine of ringed seals exposed to oil via immersion and ingestion suggests that the renal system may be active in the depuration of petroleum hydrocarbons and their metabolites. Harp seal pups fed up to 75 mL of crude oil (far in excess of that calculated to have been swallowed by the above ringed seals) showed no clinical, biochemical or morphological evidence of tissue damage (Geraci and Smith 1976). The most significant pathological effect of oil exposure observed in seals to date appears to be the increase in eye irritations (Section 2.3.1.3). Conclusions from other investigations of hydrocarbon uptake and depuration related to possible tissue damage are summarized in the following section.

2.3.1.7 Hydrocarbon Uptake, Transformation and Depuration

Pinnipeds exposed to an oil spill may inhale volatile petroleum hydrocarbons, ingest oil during nursing, or absorb oil through the skin or respiratory surfaces. An understanding of the uptake, distribution and metabolism of hydrocarbons is necessary to interprete physiological and clinical effects of oil exposure in seals.

The effects of ingested petroleum hydrocarbons on earless seals were examined by Geraci and Smith (1976). During a 24-h exposure to Norman Wells crude oil, ringed seals accumulated petroleum hydrocarbons in blubber, brain, muscle, kidney and liver. However, the highest hydrocarbon levels were detected in the bile and urine, suggesting that the kidney and liver are principal sites of detoxification and excretion (Engelhardt et al. 1977).

Two oil ingestion studies with ringed seals were designed to examine the fate and effects of hydrocarbons absorbed from the gut when test animals were fed fish containing a dose of 5 mL of isotopically-labelled crude oil per day for 5 days (Engelhardt 1978). Only low levels of oil remained in the blood, plasma, blubber, liver and muscle 24 days after the end of the oil ingestion period (Engelhardt et al. 1977), suggesting that any petroleum hydrocarbons accumulated during the treatments were largely depurated over this period. Two days following the appearance of petroleum hydrocarbons in the tissues, the bile and urine contained high levels of hydrocarbons or their metabolites (Engelhardt et al. 1977; Engelhardt 1982). Both renal and biliary excretion mechanisms appeared to be effective in clearing the blood and most tissues of the accumulated hydrocarbons and their subsequent metabolites by 7 days.

The amount of oil administered in this study was considered representative of the upper limit of the quantity of oil that a seal would likely ingest from live, oil-contaminated prey. The authors suggested that relatively large doses of oil ingested over a short period probably would not cause any acute organ or tissue damage in ringed and harp seals (Geraci and St. Aubin 1980). However, longer exposures to oil would probably result in increased hydrocarbon levels in tissues, particularly in blubber (Kooyman et al. 1976).

Geraci and Smith (1976) also calculated that ringed seals exposed to Normal Wells crude oil for 24 h absorbed some volatile hydrocarbons through the respiratory tract. Nevertheless, lung damage was not apparent even though the design of the experimental holding pen resulted in exposure to higher concentrations of volatile fractions than would normally be encountered during an actual oil spill. The effects of prolonged inhalation of volatile petroleum hydrocarbons have not been documented, although these fractions are rapidly lost from surface oil slicks due to evaporation, decreasing the potential for long-term inhalation of hydrocarbons by seals or other marine mammals.

2.3.1.8 Effects of Oil Spill Countermeasures

The effects of exposure of pinnipeds to oil spill dispersants or D/O mixtures have not been examined. However, it is possible that the natural skin oils coating the pelage of fur seals could absorb dispersants directly from the water column. This in turn could increase the wettability of the fur and alter its thermal conductivity. The extent to which changes in the thermal regime might affect survival of exposed individuals would likely depend on several factors including air and water temperature (ESL 1982a; Smiley 1982).

The chemical toxicity of D/O mixtures to pinnipeds has not been documented. The external irritation of eyes and nostrils that has been reported with crude and refined oils would probably be less severe with chemically dispersed oil because dispersant-coated oil droplets have a much lower tendency to stick to foreign materials (Canevari 1977, 1978). However, the effects of ingestion of D/O mixtures are unknown.

2.3.1.9 Indirect Effects

Indirect effects of oil exposure on pinnipeds may include spatial or temporal displacement from critical or important habitats, increased susceptibility to predators, and local reduction in food supplies or changes in normal behaviour caused by disturbances associated with cleanup operations. For example, contamination of benthic prey organisms and sediments with petroleum hydrocarbons could have indirect effects on pinnipeds which feed in benthic environments (e.g. bearded seal, walrus). Since natural stress may be a major factor influencing sensitivity to oil spills, populations near the carrying capacity of their environment may be more seriously affected by both direct and indirect impacts of oil spills. In general, these potential effects are poorly documented because they can only be studied under experimental field or actual spill situations.

2.3.2 Vulnerability of Pinnipeds to Marine Oil Spills

Pinnipeds occupy a wide range of marine habitats including shorelines, sea-ice, surface waters, the water column and benthic environments. Some species considered in this review are found exclusively in the Arctic (e.g. ringed seals, bearded seals), while others range between temperate and subarctic waters (e.g. fur seal, sea lion). The potential for pinnipeds to contact oil in the event of a marine spill varies among species, and would depend on the time of the event, the extent and location of areas contaminated, and species present in the affected areas. Nevertheless, the probability of repeated oil contact is probably greater than for other marine mammals because (1) spilled oil tends to collect on or near shorelines or ice, and (2) some species of pinnipeds occupy these habitats during relatively critical life history stages including mating, pupping and moulting. While it is not known if pinnipeds can detect oil, there is little evidence to suggest that they actively avoid it. The behavioural stimuli to nurse pups, mate or haulout may overwhelm the response to avoid oil, and this could result in repeated contamination of some individuals.

The effects of oil exposure on seals have been the subject of numerous laboratory and field investigations, and as discussed in previous sections, contact with oil can lead to a range of physical, behavioural, physiological and indirect effects. Oil is believed to have its most pronounced effect on thermoregulation and metabolism in species which rely on fur for insulation. For this reason, the northern fur seal is considered particularly vulnerable to marine oil spills.

Several species of hair seals are also considered vulnerable to oil spills as a result of the habitats they occupy and their activities in these habitats. For example, the ringed seal, an important component of arctic marine ecosystems and a species harvested by residents of virtually all coastal Arctic communities, has periods of particular vulnerability to oil spills. Pups are born in subnivean birth lairs, the lactation period is lengthy (i.e. 6 weeks), and pupping is followed by a month-long period when ringed seals haul out on the ice to moult. In the event of an oil spill, inhalation of volatile compounds and coating of pelage could cause serious damage to animals in lairs, while repeated contact with oil may occur during the moult if seals repeatedly enter contaminated waters.

Populations of other pinnipeds are vulnerable to oil spills because of specialized feeding habits or colonial breeding behaviour which increase the potential for contact with oil. For example, the walrus, fur seal and sea lion are vulnerable to spills in restricted areas used as traditional colonies. Walruses and bearded seals are also vulnerable to repeated oil exposure because they feed extensively in benthic habitats where oil could persist for extended periods.

SECTION 2.4

WHALES, DOLPHINS AND PORPOISES

Linda Martin

There have been relatively few investigations of the effects of oil on cetaceans, and these have been limited to recent studies involving bottle-nose dolphins (*Tursiops truncatus*) and to a more limited extent bowhead, fin and gray whales (*Balaena mysticetus*, *Balaenoptera physalus* and *Eschrichtius robustus*, respectively). There are no available data describing the effects of oil on any other species of whales or porpoises that occur in cold marine waters. The results of experimental studies have been recently reviewed by Thomson and McLaren (1982) and Richardson et al. (1983), while Duval et al. (1981) summarized the available information regarding the effects of past oil spills on these species. In addition, the effects of oil on this important group of marine mammals, which includes some endangered species, were summarized in recent Environmental Impact Statements prepared for offshore hydrocarbon developments in Canadian and Alaskan waters, including the Beaufort Sea-Mackenzie Delta EIS (Dome Petroleum et al. 1982) and the Gulf of Alaska/Cook Inlet EIS (U.S. Dept. of the Interior 1984).

Cetaceans occupy surface waters for relatively limited periods of time to breathe and (in some species) to feed, but the majority of their activities occur within the water column. However, only the effects of surface oil on cetaceans are described in the literature. Therefore these marine mammals are discussed in this chapter, rather than in Chapter 4.0 which describes the effects of oil spills on species in pelagic habitats.

2.4.1 Sensitivity to Oil

2.4.1.1 Mortality

There is no evidence that mortality of cetaceans has resulted from oil spills in cold marine waters (ESL 1982a), although OSIR (1983, cited in U.S. Dept. of the Interior 1984) indicated that dugongs (sea cows) died due to respiratory stress as a result of inhaling oil following a large spill in Saudi Arabia. Whether these animals were concentrated in an area contaminated by oil, or if the warmer temperatures in the region contributed to the deaths is not clear from the available information. The fact that mortality of cetaceans has not been documented following oil spills is consistent with incidental observations of whales and dolphins swimming and feeding in oil slicks (Goodale et al. 1979; Gruber 1980, cited in U.S. Dept. of the Interior 1984), as well as with the results of behavioural studies completed by Geraci and St. Aubin (1982) demonstrating that gray whales will swim through slicks resulting from natural oil seeps. While mortality of cetaceans in cold marine waters due to effects of oil has not been observed to date, in a summary of current knowledge regarding the potential lethal effects of oil spills on these mammals, Geraci and St. Aubin (1982) suggested that (1) whales trapped in oil-contaminated areas such as leads in ice or enclosed bays may be killed by oil, and (2) animals already stressed by disease or other health problems are more likely to be affected by oil.

2.4.1.2 Behavioural Effects

The only investigations of the effects of oil on the behaviour of cetaceans were those of Geraci and St. Aubin (1982), and included experimental studies with the bottlenose dolphin and field observations of gray whales in areas of natural oil seeps. The visual and echolocation capabilites of bottlenose dolphins were tested with a variety of petroleum substances under different exposure conditions. In optimum conditions of light and water clarity, trained dolphins were capable of visually detecting substances with an optical density greater than 0.05. Geraci et al. (1983) also estimated that "naive" dolphins could detect oil films having an optical density greater than 0.20 to 0.34. The precise thickness of a film of this optical density was not determined, but it was expected to be less than 1 mm. The thinnest crude oil films tested were 1-mm thick and had an optical density greater than 3. These authors concluded that dolphins could visually detect weathered surface oil in the form of pancakes, but that thin sheens (2-30 µm) that typically form following spills of light refined products (e.g. gasoline and diesel fuel) would not be easily detected.

Blindfolded dolphins were tested for their ability to detect petroleum by echolocation. Behavioural observations indicated that test animals could detect 12-mm thick patches of some crude oils and Bunker C, and that their detection capabilities increased when air bubbles were introduced to the slicks (Geraci et al. 1983). However, 6-mm thick films of crude and diesel oil were not detected. Since test animals were capable of detecting petroleum, these researchers also investigated the ability of bottlenose dolphins to avoid surface oil. Under the conditions of the experiment, dolphins avoided mineral oil by diving abruptly on contact with the surface slick and then returning to unoiled areas within the enclosure. Smith et al. (1983) reported that after initial contact with innocuous mineral oil, dolphins avoided swimming under the slick during subsequent exposures. This suggests that oil slicks from actual spills may act as a barrier to dolphins. However, Geraci and St. Aubin (1982) also emphasized that the behaviour observed in experimental situations may be modified in the natural habitat by a variety of factors including social interactions and adverse light and water conditions.

Several responses by gray whales to surface oil were observed in an area of natural oil seeps off California (Geraci and St. Aubin 1982). These included: (1) a decrease in the duration of time spent at the surface and in the frequency of blows in oil-contaminated areas relative to those in oil-free areas; (2) faster blow rates by animals in oiled areas; (3) altered swimming speeds in oiled areas, although a consistent pattern was not apparent; and (4) occasional radical changes in direction of movement by whales when approaching oil. However, Geraci and St. Aubin (1982) cautioned against extrapolation of all the observed behaviour of these whales to actual oil spill events, since it was not possible to follow specific whales into and out of oil-contaminated areas.

2.4.1.3 Physical Effects

The limited available data regarding the physical effects of oil on cetaceans are also largely from investigations by Geraci and St. Aubin (1982). These authors examined the effects of oil on the skin of bottlenose dolphins and a sperm whale (Physeter catodon), and on the baleen of mysticete whales. The skin of bottlenose dolphins exposed to crude oil and lead-free gasoline for 1.25 h changed colour slightly, but showed no evidence of damage or loss of integrity and returned to normal colour within 2 h. Geraci and St. Aubin (1982) contrasted these results to those with human skin, which required up to 7 months to return to normal coloration after similar exposure. Histological and ultrastructural examination of affected cells also indicated that recovery was rapid and occurred within 3 to 7 days. There were no differences in the healing rates of oil-contaminated wounds compared to uncontaminated incisions. Basal layers and the underlying dermis of a live-stranded sperm whale were unaffected by 17-h exposure to crude oil and gasoline applied to the surface of the skin, and the authors suggested that this was evidence of the resiliency of cetacean skin. The only other study of the physical effects of crude oil on the skin of cetaceans involved use of preserved skin from a bowhead whale. Although possible structural effects of the oil on bowhead skin were not examined, Albert (1981, cited in U.S. Dept. of Interior 1984) did report that crude oil adhered to the skin, particularly in roughened areas.

Oil may foul the baleen of mysticete whales and impair feeding, perhaps by causing structural changes in the baleen fibers. Geraci and

St. Aubin (1982) monitored water flow through fin and gray whale baleen plates before and after contamination with three types of oil. Light- to medium-weight oils caused transient (<1 min) changes in water flow rates, while heavier oil (Bunker C) reduced flow rates for up to 15 min. Oil persisted in the baleen fibers for 1-20 h after water flow rates had returned to normal, with the period of persistence increasing with the viscosity of the oil. Research on bowhead whale (Balaena mysticetus) baleen indicates that longer-term decreases in water filtration rates may occur in this species after oil contamination (Braithwaite 1983, cited in U.S. Dept. of Interior 1984). Average filtering efficiencies of baleen decreased by 5.9 to 11.3 percent after oiling, and while effects persisted for as long as 30 days, filtering efficiencies began to return to normal 8 h after fouling. Contamination of the baleen of mysticete whales with oil would likely affect feeding efficiency, and could affect energy balance if reductions in food intake were of sufficient magnitude and duration.

The effects of oil on the structure and composition of baleen fibers could be of greater long-term concern due to the potential for prolonged reductions in feeding rates. Geraci and St. Aubin (1982) examined fin and gray whale baleen plates soaked in gasoline, crude oil and tar for exposure periods exceeding those which might be expected under natural conditions. Oil affected several components of the baleen including α-keratin, lipids and trace elements. However, changes in these compounds did not affect tensile properties or x-ray diffraction characteristics of the baleen.

Oil exposure may also cause eye irritation in cetaceans. Although eye irritations were not observed during the investigations of Geraci and his co-workers, the irritating properties of oil on ocular and perocular tissues of seals have been well documented (Geraci and Smith 1976).

2.4.1.4 Effects on Growth, Development and Reproduction

There has been no research that has directly examined the effects of oil on the growth, development or reproduction of cetaceans. However, the potential exists for effects on these processes due to ingestion of oil or oil-contaminated prey and reductions in feeding rate.

2.4.1.5 Physiological and Pathological Effects

Other than the examination of biochemical processes in epidermal cells conducted by Geraci and St. Aubin (1982), the physiological and pathological effects of oil exposure in cetaceans have also not been directly investigated. These researchers examined four cell processes, but only phospholipid synthesis showed a consistent reduction with oil exposure. Phospholipid production is essential for maintenance of cell membrane structure and stability.

Respiratory stress could result from exposure to petroleum vapours if surface oil was contacted during the breathing cycle of cetaceans. The potential effects of inhalation of petroleum vapours are discussed by Geraci and St. Aubin (1982). The effects of inhalation of petroleum vapours would depend on a number of factors including the duration of exposure, sensitivity of the animal due to other stresses, and hydrocarbon concentrations in vapours at the water surface. However, Geraci and St. Aubin (1982) suggest that only whales or dolphins unable to leave the immediate area of a spill, or confined to a contaminated lead or bay would likely be affected by petroleum vapours, since oil that has weathered in the open ocean for as little as 2 to 4 h has lost all volatile components. In addition, these authors suggest that effects due to direct inhalation of oil are unlikely because the breathing mechanism of cetaceans prevents the inhalation of water and should prevent direct uptake of oil in the same manner.

2.4.1.6 Hydrocarbon Uptake, Transformation and Depuration

There is little published literature on the metabolic fate of ingested oil or oil-contaminated prey in cetaceans. The only report of oil ingestion linked with the death of a cetacean is that of Dugay (1978, cited in U.S. Dept. of Interior 1984), who reported the presence of petroleum hydrocarbons in the intestine of a stranded bottlenose dolphin. However, there was no evidence to suggest that oil ingestion was responsible for the stranding and subsequent death of this animal. The only experimental study which has directly examined the effects of ingested oil on cetaceans was completed by Caldwell and Caldwell (1981). Hydraulic oil was concealed in capsules and fed to a bottlenose dolphin over a 3-month period. No clinically detectable effects attributable to the oil were evident during autopsy of the dolphin after the experiment.

Although Geraci and St. Aubin (1982) did not complete oil ingestion studies, the volume of fuel oil that would be acutely toxic to a cetacean was calculated on the basis of critical oil doses established for rats. They concluded that cetaceans such as dolphins which drink relatively small quantities of water (less than 1.5 L day^{-1}) would be unlikely to consume sufficient quantities of oil to cause acute toxicity (5-25 mL kg^{-1}). However, ingestion of oil-contaminated prey by cetaceans could result in accumulation of hydrocarbons in body tissues. Benthic feeders such as gray whales would be most likely to consume oil-contaminated prey.

There is some indirect evidence that cetaceans may accumulate hydrocarbons from ingested oil and may have the ability to detoxify at least some of these compounds. Geraci and St. Aubin (1982) examined 258 tissue samples from marine mammals stranded along the Atlantic coast of Canada and the United States to determine hydrocarbon levels. Naphthalene was found in most tissues analyzed, with the highest concentrations occurring in blubber. Geraci and St. Aubin (1982) suggested that mobilization of fat stores, which might occur during migration or lactation, would produce transiently higher levels of naphthalene in the blood and simultaneously increase their availability for detoxification and subsequent excretion. During the same investigation, analyses of liver samples indicated the presence of low to moderate concentrations of cytochrome P450, an enzyme in the mixed function oxidase group that is active in the detoxification of petroleum hydrocarbons in other mammals and fish. Further research is required to determine if this enzyme is involved in the detoxification of petroleum hydrocarbons by cetaceans, and if its production stimulated by the presence of hydrocarbons.

2.4.1.7 Effects of Countermeasures

There are no published data describing the effects of chemical dispersants, dispersed oil mixtures or other oil spill countermeasures on cetaceans. Physical removal of surface oil slicks would minimize dispersion of oil into the water column, and would also minimize the potential for contact by cetaceans. Smiley (1982) suggests that application of chemical dispersants may minimize effects of oil spills on marine mammals when both floating oil and chemical dispersant are effectively diffused into the water column. While this countermeasure would reduce the risk of direct contact with surface slicks, it would also increase the concentrations of dispersed oil and dissolved hydrocarbons in the water column. The possible effects of dispersed oil clouds and short-term increases in the concentrations of dissolved hydrocarbons in the water column on cetaceans remain unknown although exposures would be of short duration.

2.4.1.8 Indirect Effects

The indirect effects of oil spills on cetaceans have not been documented, but they are expected to be largely socio-economic in nature. For example, any mortality of a species on the endangered species list (e.g. bowhead whale) would likely have serious socio-political ramifications, while either the mortality of harvested species (e.g. white whale) or the inability to hunt such species due to conflicts with oil spill cleanup responses would almost certainly have socio-economic implications.

2.4.2 Vulnerability

In general, cetaceans occupy pelagic habitats more than surface waters and, therefore, are potentially more vulnerable to oil present in the water column. However, studies of the fate and effects of oil on cetaceans have dealt primarily with the effects of surface oil slicks. There have been no studies of the effects of chemical dispersants or of dissolved or dispersed oil in the water column on cetaceans. However, vulnerability to oil present within the water column would be relatively short-term because hydrocarbon concentrations in the water column beneath oil slicks are typically very low within a few hours of spills.

As indicated earlier, cetaceans would likely be most vulnerable to surface slicks when already stressed by other natural factors such as disease or confined in areas containing oil (e.g. polynyas, leads and embayments). The research of Geraci and St. Aubin (1982) indicates that cetaceans may have the ability to detect oil slicks. Detection and avoidance of a spill area that restricted their migratory movements or their use of preferred feeding areas are possibly the most significant concerns.

CHAPTER 3

OIL WITHIN THE WATER COLUMN

The following sections summarize available information concerning the biological fate and effects of petroleum hydrocarbons on organisms that occur within the water column. Separate sections address the sensitivity and vulnerability of phytoplankton, zooplankton and fishes to oil spills. As in other chapters of this report, it is important to stress that some marine flora and fauna may be exposed to oil in more than one habitat. In shallow areas, phytoplankton may be found throughout the water column. Zooplankton may also occur to great depths in some marine waters, particularly those species that undergo extensive diel vertical migrations. Some species of fish have buoyant eggs that float on the water surface where they could be exposed to surface slicks, while the eggs of other species are distributed widely within the water column. Still other species deposit their eggs on or within benthic sediments. A wide variety of demersal fish species are found in close association with the seafloor as adults and may contact oil reaching benthic habitats or ingest oil-contaminated benthic prey organisms.

Conversely, it is emphasized that biota discussed in other chapters may likewise be affected by oil within the water column. As indicated in the previous chapter, marine mammals spend the greatest portion of their time within the water column rather than at the water surface, and therefore could be exposed to both dissolved and dispersed oil. Similarly, several species of birds are pelagic feeders and therefore may contact oil within the water column as well as on the water surface. Benthic organisms (discussed in Chapter 4) may also be affected by oil dissolved or dispersed in the water column.

SECTION 3.1

OVERVIEW OF PHYSICAL AND CHEMICAL PROCESSES

Wayne Duval

The behaviour and fate of surface oil slicks were briefly reviewed in Section 2.1. The present section provides a comparable overview of the dominant physical-chemical processes that occur once oil is dispersed into the water column. The influence of chemical dispersants on these processes is also briefly discussed. A detailed review of the fate and behaviour of oil in cold climates is provided in Mackay (1984). Figure 3.1-1, taken from this source, illustrates the major physical, chemical and biological processes that occur within the water column, as well as in surface slicks and benthic sediments. As evident from this figure, a relatively large number of processes affect the fate of oil within the water column, although only those which have biological implications are described here.

3.1.1 Dissolution

The hydrocarbons present in crude oils and refined petroleum products are only slightly soluble in seawater. Their average solubility is about 30 g m^{-3} and, therefore, does not contribute significantly to mass loss following an oil spill (Mackay 1984). However, this process is exceedingly important biologically because it is largely dissolved hydrocarbons that are the source of impact of oil spills on most organisms present in the water column. The most soluble hydrocarbons are the low molecular weight aromatics. These tend to be the most acutely toxic, but are also volatile and therefore lost very rapidly due to evaporation from surface slicks. Dissolution of hydrocarbons occurs from both surface slicks and oil subsequently dispersed within the water column.

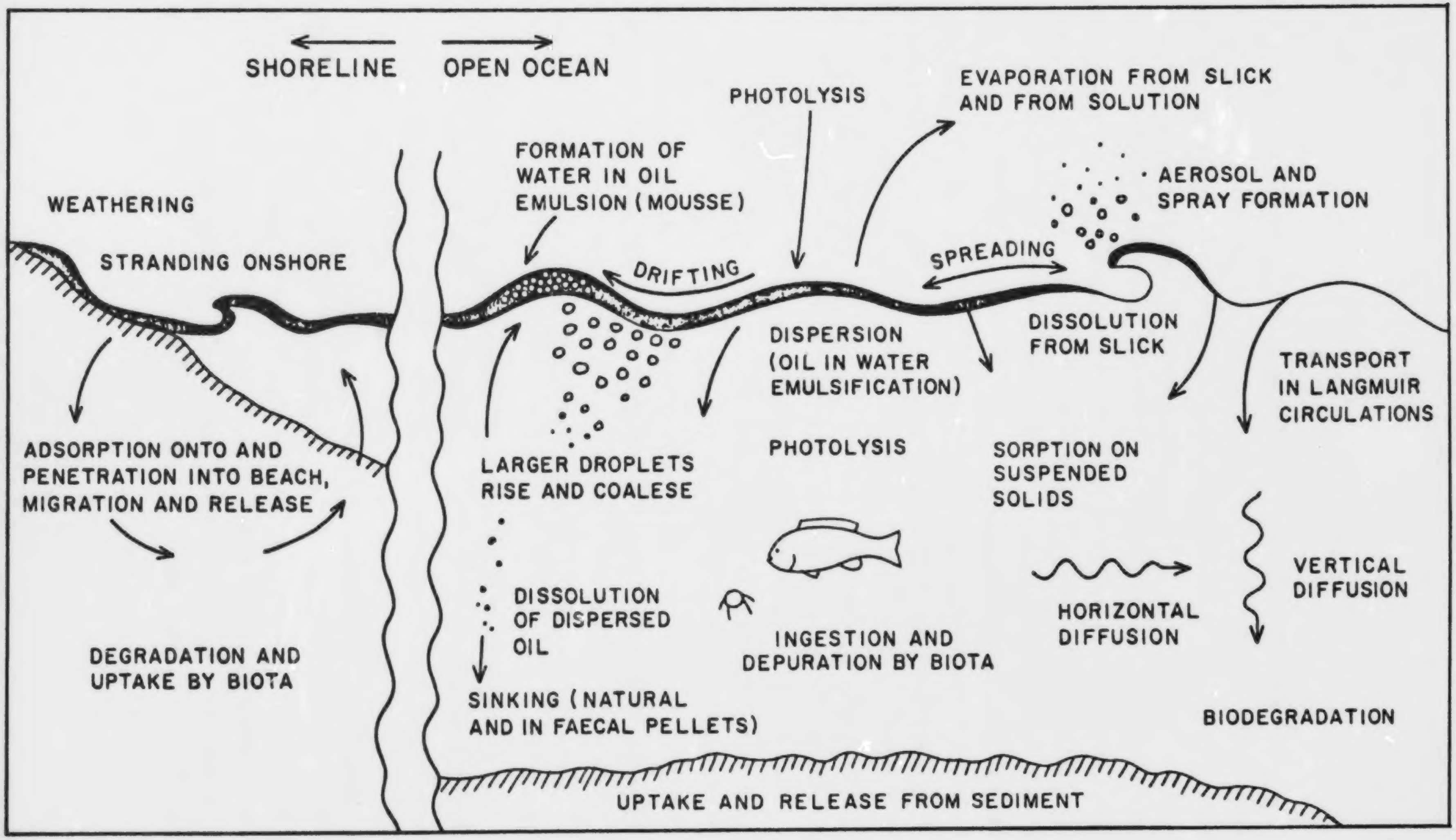
SHORELINE
OPEN OCEAN
WEATHERING
STRANDING ONSHORE
FORMATION OF WATER IN OIL EMULSION (MOUSSE)
PHOTOLYSIS
EVAPORATION FROM SLICK AND FROM SOLUTION
AEROSOL AND SPRAY FORMATION
DRIFTING
SPREADING
ADSORPTION ONTO AND PENETRATION INTO BEACH, MIGRATION AND RELEASE
LARGER DROPLETS RISE AND COALESE
DISPERSION (OIL IN WATER EMULSIFICATION)
DISSOLUTION FROM SLICK
TRANSPORT IN LANGMUIR CIRCULATIONS
PHOTOLYSIS
SORPTION ON SUSPENDED SOLIDS
VERTICAL DIFFUSION
HORIZONTAL DIFFUSION
DISSOLUTION OF DISPERSED OIL
INGESTION AND DEPURATION BY BIOTA
DEGRADATION AND UPTAKE BY BIOTA
SINKING (NATURAL AND IN FAECAL PELLETS)
BIODEGRADATION
UPTAKE AND RELEASE FROM SEDIMENT

Figure 3.1-1 Oil Spill Processes (Source: Mackay 1984)

The rate of dissolution of hydrocarbons from oil dispersed in the water column increases with the square of the diameter of oil particles (Mackay 1984). This relationship is extemely important with respect to the potential effects of application of chemical dispersants on oil concentrations within the water column and subsequent effects on pelagic flora and fauna. Surface oil slicks that are naturally dispersed into the water column through turbulent processes at the seawater-air interface (e.g. wave action) tend to result in the formation of relatively large oil droplets (0.1-1.0 mm). On the other hand, the application of chemical dispersants to surface oil slicks produces an oil-in-water dispersion containing substantially smaller oil droplets (e.g. 10 μm). Any soluble hydrocarbons present in droplets of this size would completely dissolve within a few minutes or hours (Mackay 1984) due to the large oil surface area:volume ratio in the dispersed oil cloud. It is important to recognize the biological implications of this phenomenon. Where oil is naturally dispersed within the water column, the dissolution of hydrocarbons may proceed at a slower rate but over a longer period, and pelagic organisms will generally be exposed to lower hydrocarbon concentrations because of the natural dilution of these compounds in surrounding waters. However, pelagic organisms may be exposed to much higher hydrocarbon concentrations when oil is chemically dispersed because the soluble components tend to dissolve quickly and therefore have less opportunity to be diluted in adjacent waters.

The overall concentration of hydrocarbons in the water column is determined by the amount of soluble hydrocarbons present in the spilled oil and the water-oil volumetric ratio. For example, if 10 percent of the oil is potentially soluble at a water-to-oil ratio of 100 000, the maximum concentration of hydrocarbons in the water column would be 1 part per million (Mackay 1984). Concentrations of dissolved oil in the water column following oil spills and blowouts have rarely been measured (particulate or whole oil has been measured more frequently). The data provided in Kanter et al. (1983) and summarized in Table 3.1-1 show that concentrations of dissolved hydrocarbons in the water column following a crude oil spill in the

TABLE 3.1-1

SUMMARY OF HYDROCARBON CONCENTRATIONS MEASURED IN THE WATER COLUMN FOLLOWING OIL SPILLS AND BLOWOUTS

(from Kanter et al. 1983)

Spill Event	Location	Volume/ Type of Oil	Sample Type	Hydrocarbon Conc. (ppb)
ARROW	Chedabucto Bay Nova Scotia	68,000 bbl/ Bunker C	Particulate Whole Water	16-40 2-131
Ekofisk	North Sea	12,700 tons crude/ 19,000 tons gas	Whole Whole	100-400 20-40
Platform MPB Chevron	Mississippi Delta	9,000 tons/ crude	Dissolved	200
Refinery	Honshu, Japan	40,000 tons/ Bunker C	Whole	900-43,700
Refinery	Japan	4,000 tons/ Marine Diesel	Whole	45-60
ARGO MERCHANT	Mass.	29,000 tons/ No. 5 Fuel Oil	Whole	90-340
TSESIS	Sweden	17,500 tons/ No. 5 Fuel Oil	Whole	51-58
KURDISTAN	Nova Scotia	7,500 tons/ Bunker C	Whole	0.5-5
AMOCO CADIZ	France	220,000 tons/ Arabian crude	Whole Whole	5-45 26-330
Ixtoc	Gulf of Mexico	476,000 tons/ crude	Whole Dissolved	100-10,000 150-200

Mississippi Delta and during the Ixtoc I blowout in the Gulf of Mexico were in the range 150-200 ppb.

3.1.2 Chemical and Biochemical Conversion

Hydrocarbons may be oxidized through two processes (1) photolysis or photochemically-induced oxidation, and (2) biodegradation or enzymatically-induced oxidation. Photolysis occurs at the air-sea interface, but affects organisms in the water column. Light absorption by oil triggers a series of complex reactions that result in a range of oxygenated products including alcohols, aldehydes, ketones, carboxylic acids, hydroperoxides and phenols (Mackay 1984). Because these compounds are more polar than their hydrocarbon precursors, they tend to be water-soluble and dissolve more rapidly. Not only are many of these oxidation products toxic to pelagic flora and fauna, but Tjessem et al. (1984, cited in Mackay 1984) suggested that these compounds may be sufficiently toxic to inhibit microbial degradation for a period of weeks.

The second oxidation process in the water column is biodegradation by various micro-organisms. Biodegradation is probably the most significant long-term mechanism of hydrocarbon detoxification in the marine environment (Mackay 1984), and is discussed in more detail in Section 4.5 of this report.

3.1.3 Sedimentation of Oil

When suspended sediments are present in the water column, dissolved hydrocarbons may be sorbed onto the surface of particles and then sink to the seafloor. Although clays and other inorganic materials have some sorptive capacity, it is widely accepted that the organic carbon content of suspended sediments dominate the sorption behaviour (Mackay 1984). In addition, a second process involving the direct association or flocculation of oil particles and suspended sediments may result in a combined particle with sufficient density to sink. This process depends on the relative diameter and volume of oil and sediment particles. Mackay (1984) reports that if an oil drop with a density of 0.85 g cm^{-3} were twice the diameter (8.3 times the volume) of the suspended sediment particle, the combined particle would be equal in density to seawater. Therefore, at lower ratios the particle will sink and at higher ratios the particle will be buoyant.

Zooplankton may also assume a significant role in the sedimentation of particulate oil in the water column. Sedimentation involves the encapsulation of oil droplets in fecal pellets of these organisms, and is discussed in detail later in this chapter (Section 3.3.2).

SECTION 3.2

PHYTOPLANKTON

Ken Trudel

The interactions of petroleum hydrocarbons with phytoplankton are of interest for three reasons. First, phytoplankton are the primary producers in pelagic communities, so that oil spills could locally affect the community energy source. Secondly, phytoplankton represent a potentially important entry point for hydrocarbons into marine food chains. Finally, although planktonic algae form only a small proportion of the total biomass of a pelagic community, they form a large surface area for the adsorption and absorption of hydrocarbons in the water column because of their large surface-to-volume ratio, and hence may significantly affect the fate of hydrocarbons entering marine environments. As with other marine organisms, some species of phytoplankton are more sensitive than others to the effects of oil spills. However, unlike most other groups, the effects of spills on phytoplankton communities are very localized and short-lived. This is because the affected phytoplankton standing stock and primary production quickly return to prespill levels through the natural transport of organisms from surrounding areas and through the rapid reproduction of phytoplankton populations.

Oil-phytoplankton interactions have been reviewed by several authors (O'Brien and Dixon 1976; Vandermuelen and Ahern 1976; Corner 1978; Laperriere 1980; Van Baalen 1981; Curbs 1982; Snow 1982; Gaur and Rai 1983). In the present review, the state of knowledge regarding petroleum-phytoplankton interactions in general, and in cold climates in particular, is discussed relying on the results of laboratory research, mesoscale investigations, and observations from field tests and actual spills.

3.2.1 Laboratory Culture Studies

Various researchers have used unialgal cultures (both batch and chemostat cultures) to study the effects of petroleum hydrocarbons on planktonic algae. Unlike ecotoxicological studies with most other marine flora and fauna, death of the test organisms is seldom used as a parameter or end point in these studies. Physiological indices are used instead including rates of respiration and nitrogen fixation, protein or DNA synthesis, and, most commonly, rates of photosynthesis or population growth. The following sections discuss (1) the concentrations of hydrocarbons in the water column necessary to affect phytoplankton populations and the factors that influence observed responses, and (2) the mechanism by which petroleum hydrocarbons cause phytotoxicity.

3.2.1.1 Mechanism of Phytotoxicity

Despite several decades of research, the mechanism of phytotoxicity remains somewhat obscure. However, it is apparent that it is related to the chemotoxic effects of dissolved hydrocarbons. Studies of terrestrial plants have suggested that petroleum hydrocarbons exert their primary effect through disruption of the structure and function of the plasma membrane of cells by displacing the natural lipid constituents. Baker (1970) suggested that similar disruptions might occur in the plasma membrane of phytoplankton and in the membranes of cell organelles. This theory is consistent with the observations of Kraus et al. (1973), who reported that for a range of aromatic hydrocarbons, toxicity increased with increasing lipid solubility and hence with the affinity of hydrocarbon molecules for the plasma membrane. Further evidence for this mechanism is seen in the work of Hutchinson et al. (1979) who found that potassium and manganese leakage from algal cells was increased by exposure to hydrocarbons. Such a disruption of the plasma membrane would reduce its effectiveness as a semi-permeable membrane, and thereby decrease overall cell viability. The disruption of the membranes of cell organelles such as mitochondria could also affect the membrane-associated enzyme systems involved in oxidative phosphorylation, and aerobic respiration would be disrupted (Baker 1970). O'Brien and Dixon (1976), citing Manwell and Baker (1967), suggested that petroleum hydrocarbons might also influence photosynthesis by disrupting the normal energy transfer along the light absorption pathway involved in the process.

3.2.1.2 Sensitivity of Planktonic Algae to Oil

Considerable research has focussed on elucidatation of the relationships between ambient hydrocarbon concentrations and the inhibition of population growth and photosynthesis in laboratory cultures of phytoplankton. Unfortunately, much of the early work is difficult to interpret since oil concentrations in the exposure medium were not measured. Van Baalen (1981) summarized the results of a number of later studies in which hydrocarbon concentrations were measured (Gordon and Prouse 1973; Pulich et al. 1974; Soto et al. 1975; Parsons et al. 1976), and concluded that the lowest level at which inhibition of photosynthesis might occur would be 20-40 ppb of hydrocarbons in sea water. Similar concentrations were shown to inhibit carbon fixation during *in situ* studies of natural phytoplankton communities.

There is some variability in the sensitivity of different algal species to petroleum hydrocarbons (Gaur and Kumar 1981; Hsiao 1978; Pulich et al. 1974). Most studies suggest that interspecific variation is small (Ostgaard et al. 1984), but in research involving 11 species, Mironov (1968, 1972) found that the most sensitive species showed a reduction in cell division rate at a nominal oil concentration of 0.01 ppm (as oil added), while the more resistent species were inhibited by oil concentrations of 10-100 ppm (as oil added). The actual range in sensitivity would probably have been less if dissolved hydrocarbon concentrations had been measured rather than expressed as the amount of oil added. Gaur and Rai (1983) noted that in situations involving chronic oil pollution, exposure of algal communities to oil and its products resulted in the elimination of susceptable species and a proliferation of resistant forms because of reduced competition and less grazing pressure.

Of particular interest in the Arctic, where the phytoplankton community is comprised largely of diatoms and flagellates, was the discovery that some species of diatoms appear to be more sensitive to many oils than flagellates (Hsiao 1978).

Nuzzi (1973) and others have shown that water-soluble fractions (WSF) of crude and refined oils differ in their phytotoxicity. These differences reflect variations in the relative abundance of the more toxic hydrocarbons in the parent oils and resultant WSFs. There is some evidence that the toxicity of oils is due to the low molecular weight aromatics which, although only moderately toxic, are present in relatively high concentrations in the WSFs of most oils. However, some research has shown that certain phytoplankton groups are particularly sensitive to specific compounds (Pulich et al. 1974; Winters et al. 1977). For example, Pulich et al. (1974) found that certain substituted anilines were particularly toxic to some species of blue-green algae, but were relatively innocuous to other species. Although earlier workers concluded that the toxicity of crude oils was due largely to the lighter, more volatile aromatics such as benzene and toluene, recent evidence suggests that the photo-degradation products of oils are far more toxic to phytoplankton than the parent compounds (Lacaze and Villedon de Naide 1976; Larson et al. 1979; Ostgaard et al. 1984). Several authors have examined the influence of environmental variables on oil-phytoplankton interactions. For example, Hsiao (1978) found that temperature influenced the sensitivities of certain flagellates and diatoms to hydrocarbons. He reported that the sensitivity of cultured algae to a variety of crude oils increased with increasing temperature over the range from 0-10°C. Other authors have demonstrated that rates of growth and photosynthesis were most inhibited by hydrocarbons in situations where light (Hellebust et al. 1975; Kusk 1981) and nutrients (Stoll and Guillard 1974; Karydis 1981) were sub-optimal or limiting. Because laboratory studies are generally conducted under optimal growth conditions, and since natural conditions are seldom optimal, this suggests that natural phytoplankton communities could be more sensitive to petroleum hydrocarbons than has been indicated by the results of studies with laboratory cultures.

With some algal species and oil types, photosynthesis and/or growth are stimulated at low hydrocarbon concentrations, but are depressed at higher concentrations (Gordon and Prouse 1973; Dunstan et al. 1975; Parsons et al. 1976; Hsiao 1978). Two explanations have been suggested for this stimulatory effect (1) hydrocarbons present at non-toxic levels might be utilized as an energy source by algal cells (Karydis 1979), or (2) the WSF of oils may contain compounds that either mimic or are themselves growth-regulating substances, and these compounds may promote cell division (Gordon and Prouse 1973). Gaur and Kumar (1981) observed growth stimulation in *Anacystic nidulans* cultures in the presence of crude oil, and concluded that since the hydrocarbons affected the rate of cell division but not the final standing crop of algae in the culture, the hydrocarbons were acting as growth regulators and not simply as an energy source.

3.2.1.3 Bioaccumulation and Metabolism of Hydrocarbons by Phytoplankton

Because of their small size and therefore large surface-to-volume ratio, phytoplankton present a large surface area for absorption and adsorption of dissolved hydrocarbons. This surface area is disproportionately high relative to other groups of aquatic organisms. As a result, there is a potential for phytoplankton to adsorb or absorb a significant proportion of hydrocarbons in the water column, thereby making them available for subsequent ingestion by herbivorous zooplankton and for degradation by the biochemical hydrocarbon-degrading systems of planktonic algae. Very few studies have been conducted on the bioaccumulation and degradation of hydrocarbons by phytoplankton, despite their potentially significant role in these processes. Soto et al. (1975) examined the bioaccumulation of naphthalene by *Chlamydomonas*, and demonstrated that this species did accumulate and depurate hydrocarbons. The study did not determine a steady state concentration factor or bioaccumulation rate constants for naphthalene into the algae cells, but it did establish that *Chlamydomonas* was unable to metabolize naphthalene.

Much of the earlier research on petroleum-phytoplankton interactions suggested that phytoplankton were unable to metabolize hydrocarbons. The aforementioned investigation by Soto et al. (1975) and studies by Conover (1971) and Payne (1976) all suggested that planktonic algae lacked aryl hydrocarbon hydroxylase systems. However, more recent research has shown that some micro-algae can degrade petroleum hydrocarbons such as phenol (Ellis 1977) and naphthalene (Cernglia et al. 1979, 1980).

In offshore areas, the effects of oil spills on phytoplankton will be local and transitory. The significance of phytoplankton in terms of the overall fate of hydrocarbons entering the marine environment is also expected to be very minor. However, in areas of chronic hydrocarbon contamination, phytoplankton may assume a significant role in determining the environmental dynamics of the non-volatile petroleum hydrocarbons.

3.2.2 Controlled Microcosm Experiments

Although laboratory studies with unialgal cultures are useful in determining the physiological responses of phytoplankton to petroleum hydrocarbons, it is difficult to directly extrapolate the results of these investigations to natural pelagic communities. Pelagic communities are highly dynamic, with algal populations interacting with other members of the planktonic community. Accordingly, some researchers have conducted studies with pelagic communities isolated in enclosures or microcosms.

The earliest studies of this type appear to be those of Lacaze (1974) and involved the exposure of microcosms to Kuwait crude oil. Primary production in the treated enclosure showed a rapid but transitory decline relative to rates measured in the control enclosure. The decline in rates of production and subsequent recovery were attributed to the ephemeral effects of the volatile toxic fraction that was quickly lost from the treated enclosure.

Lee et al. (1977) added a WSF of No. 2 fuel oil to a microcosm to achieve a final concentration of 40 ppb. In the three weeks following the start of the experiment, blooms of diatoms, flagellated green algae, and microzooplankton were observed in the control enclosure. By comparison, diatom populations were virtually eliminated in the treated enclosure following the addition of oil, and the blooms of other algal groups were smaller than those observed in control enclosures. Dahl et al. (1983) layered Ekofisk crude oil on the surface of enclosures and achieved hydrocarbon concentrations of 1.4 ppm at a depth of 0.2 m and concentrations of 200-900 ppb in the remainder of the water column, depending on depth and the time after treatment. Phytoplankton (Clorophyta) blooms and increased carbon fixation were observed in the control enclosure but not in the treated enclosure. In agreement with Lee et al. (1977); Dahl et al. (1983) reported that diatoms and flagellates dominated the phytoplankton in both enclosures and, while both groups were reduced in abundance in the oiled enclosure, the diatom populations were more severely reduced than other species.

Vargo et al. (1982) and Oviatt et al. (1982) studied the effects on a partially isolated pelagic community of chronic oiling with the water-soluble fraction of No. 2 fuel oil. Unlike the earlier investigations, which were of relatively short duration (ca. one month), this study was continued for nine months. There was considerable variation in processes measured in the microcosm over the period of study, although the authors concluded that primary production and abundance of chlorophytes were greater in the oiled microcosm than in the untreated enclosure. The greater biomass of phytoplankton in the treated microcosm appeared to be the result of reduced grazing pressure by zooplankton and benthic herbivores.

The results of experiments involving exposure times of weeks or months clearly cannot be extrapolated to open-sea situations, where elevated hydrocarbon concentrations only persist for a few days following oil spills. They do, however, simulate the conditions associated with inshore spills, where water masses mix more slowly so that elevated hydrocarbon concentrations in the water column may persist for longer periods. In these situations, available data suggest that at higher hydrocarbon concentrations, primary production will be reduced and there will be a shift in species composition of the phytoplankton community. The greater sensitivity of diatoms to hydrocarbons observed in the laboratory algal culture investigations appears to be confirmed by the results of microcosm studies. At lower levels of oil contamination, primary production and algal biomass may be increased for reasons which are presently unclear. The hydrocarbon concentrations necessary to cause these effects or reductions in phytoplankton abundance and production appear to be highly variable.

3.2.3 Marine Spill Situations

As mentioned above, the responses of phytoplankton populations and communities to oil are short-term and localized, although the nature and magnitude of responses appear to vary with exposure circumstances. In some cases, carbon fixation and algal standing crops have been reduced during actual spills. For example, following the spill from the AMOCO CADIZ, the abundance of chlorophytes was reduced for several days in the vicinity of the wreck and in adjacent inshore areas (Cabioch et al. 1981). On the other hand, after the TSESIS spill and the Ekofisk blowout, primary production in the vicinity of these spills actually increased. During the Ekofisk blowout, rates of primary production in areas affected by oil were as much as 2-3 times higher than the rates measured in the surrounding waters (Lannergren 1978). The cause of this difference in the effects of different spills appears to involve interactions among the oil, phytoplankton, and the zooplankton that graze on phytoplankton. At high levels of contamination such as those documented during the AMOCO CADIZ incident, phytoplankton and zooplankton standing stocks were reduced, and resulted in a decrease in the rate of primary production. On the other hand, the increased rate of carbon fixation following the Ekofisk and TSESIS spills was probably caused by differences in the sensitivity of phytoplankton and zooplankton to oil. In both of these events, grazing pressure by zooplankton was reduced either through a reduction in the number of herbivorous zooplankton or a reduction in grazing rates, thereby encouraging increased standing crops of phytoplankton and increased primary production (Lannergren 1978; Johansson 1980).

The visible effects of past oil spills on phytoplankton populations have been extremely localized. Studies conducted by Lannergren (1978) in areas surrounding the Ekofisk blowout in the North Sea indicated that carbon fixation rates were 2-3 times higher than normal in surface waters within 2-3 km of the blowout site, where hydrocarbon concentrations were 200-300 ppb. Carbon fixation was still higher than in control areas at distances up to 10 km from the blowout site, but were not significantly different from background rates approximately 30 km from the wellhead.

Changes in the species composition of phytoplankton communities have also been observed in waters affected by oil spills (Duval et al. 1981), although in contrast to the results of laboratory and microcosm studies, there are only limited data describing changes in phytoplankton populations after actual spills. These changes are partly due to the differential sensitivity of various phytoplankton species to petroleum hydrocarbons. Nevertheless, as indicated earlier, any effects of oil spills on the species composition of phytoplankton communities would be extremely localized and short-term due to the high reproductive rates of these populations and the opportunities for recruitment of organisms from the region outside the areas affected by a spill.

3.2.4 Effects of Dispersants on Petroleum-Phytoplankton Interactions

The effects of chemical dispersants and dispersant/oil (D/O) mixtures on phytoplankton populations were recently reviewed by Curds (1982) and Snow (1982). Of all oil spill countermeasures, dispersants have the greatest potential to affect phytoplankton. Chemical dispersants substantially increase the amount of oil entering the water column, and because of the increased surface area of emulsified oil (by virtue of smaller particle diameter), the concentrations of dissolved hydrocarbons in the water column are increased after dispersant application. There is, however, no evidence that chemical dispersants themselves may be toxic to marine algae, at least at concentrations that could be realistically reached in the water column.

Older generation dispersants of the type used before 1975 were highly toxic to most forms of marine life (Snow 1982). For example, many of the effects of the TORREY CANYON spill were attributed in part to the application of large quantities of chemicals during the cleanup of coastal areas (Duval et al. 1981). However, dispersants meeting the acceptability guidelines for use in Canadian marine waters are substantially less toxic than the older formulations. This is largely related to the use of surfactants and carriers (e.g. alcohols) that are less toxic than comparable ingredients added to previous generations of dispersants (Tokuda and Arasaki 1977). However, modern dispersants are less effective than the older formulations in dispersing surface oil slicks (Bratbak et al. 1982).

The effects of D/O mixtures on phytoplankton communities, at ratios that would be effective in dispersing surface oil slicks, have been examined by Hsiao (1978) and Trudel (1978). The latter author investigated the effects of the mixture of the dispersant Corexit 9527 and Venezuelan crude oil on primary production of natural phytoplankton communities. At the recommended D/O ratios, the toxicity of the dispersed oil mixture was due to the oil itself. The Corexit 9527 did not contribute to the effects of the mixture on algal productivity. On the other hand, Hsiao (1978) found that the toxicity of D/O mixtures to arctic marine phytoplankton was greater than either the dispersant (Corexit 9527) or Prudhoe Bay crude oil alone. Similar trends have been observed in studies with marine invertebrates (Sections 4.4 and 6.3), and as indicated earlier, are likely due to the fact that the addition of dispersant markedly decreases the size of dispersed oil particles (Duval et al. 1980). As a result, the total surface area for dissolution of hydrocarbons from the oil is substantially increased, leading to higher dissolved hydrocarbon concentrations within the dispersed oil cloud.

SECTION 3.3

ZOOPLANKTON

Ken Trudel

The interactions of petroleum hydrocarbons with zooplankton communities are of interest because (1) zooplankton are major primary consumers in pelagic food webs and play an important role in marine trophodynamics, and (2) they may assume a significant role in the transfer of petroleum hydrocarbons to members of higher trophic levels. In addition, many species of zooplankton are indiscriminant filter-feeders and may play a role in sedimenting particulate oil from the water column.

The zooplankton community is a highly diverse assemblage, consisting of representatives from a number of phyla including protozoa, hydrozoans, ctenophores, polychaetes, molluscs and crustaceans. Fish eggs and larvae are also common members of the zooplankton community, but the effects of petroleum hydrocarbons on these fauna are discussed elsewhere (Section 3.4). The composition of zooplankton communities vary considerably with location and season, although calanoid copepods are by far the most dominant member of this community in temperate and arctic seas. For this reason, much of the following discussion deals with copepod-petroleum interactions.

3.3.1 Sensitivity of Zooplankton to Petroleum Hydrocarbons

3.3.1.1 Acute Toxicity

In a summary of information published up to 1973, Moore and Dwyer (1974) concluded that aqueous concentrations of total hydrocarbons in the range 0.1-10 mg L^{-1} would kill planktonic crustaceans. A decade later, Wells (1985) reviewed the literature on a wider variety of zooplankton and found that the range of published 96-h LC_{50} values was reasonably consistent with that suggested by Moore and Dwyer (1974). However, since the zooplankton community is such a diverse assemblage, there may be differences in the sensitivity of various taxa to oil. No single study has compared the sensitivities of a variety of species representing different taxa, although Wells (1985) summarized information from several investigations of the effects of water-soluble fractions (WSFs) of crude and refined oils on different zooplankton species. Experiments involving 1-2 day exposures of protozoans, ctenophores, mollosc embryos and larvae, and various crustaceans have produced LC_{50} values in the range 0.1-2.0 mg L^{-1} (Berdugo et al. 1977; Lee and Nicol 1977; Sekerak and Foy 1978; Hebert and Poulet 1980). These data therefore suggest that the concentrations of dissolved petroleum hydrocarbons lethal to most zooplankton range from 0.1 to 10 mg L^{-1}.

Some zooplankton appear to be relatively insensitive to petroleum hydrocarbons. For example, Percy and Mullins (1975) found that *Calanus hyperboreus* was much more resistant to oil-water dispersions than other arctic species examined. Bioassays conducted by Foy (1982) also indicated this particular species is also resistant to the water soluble fraction of crude oil. This author reported a 96-h LC_{50} of 73 mg L^{-1} with *C. hyperboreus* and Prudhoe Bay crude oil. The reason for this resistance to hydrocarbon toxicity is not known.

3.3.1.2 Sublethal Effects of Petroleum Hydrocarbons

Numerous sublethal responses to oil have been observed in zooplankton. These effects range in severity from brief, reversible changes in behaviour (Section 3.3.1.3) to depressed growth or reduced reproductive rates. Whereas the short-term and reversible sublethal effects would probably be inconsequential, particularly at the population level, those that alter reproduction and growth rates have the potential to cause more serious damage.

The reproductive potential of zooplankton is determined by the combined rates of egg production and egg viability, both of which have been shown to be affected by hydrocarbon exposure. Continuous exposure of copepods to petroleum hydrocarbons reduces fecundity (Ustach 1977; Ott et al. 1978). The latter authors found that naphthalene concentrations as low as 0.01 mg L^{-1} caused a significant decrease in the fecundity of *Eurytemora*. Berdugo et al. (1977) demonstrated that pulsed, short-duration exposures of *Eurytemora* to oil affected reproduction; exposures of < 4 h to a WSF of fuel oil containing hydrocarbon concentrations as low as 3 mg L^{-1} caused a decrease in fecundity. Cowles and Remillard (1983) found that copepod eggs were highly sensitive to the effects of hydrocarbons. They reported that 48-h exposure to hydrocarbon concentrations as low as 0.01 mg L^{-1} (WSF of Louisiana crude oil) resulted in a significant reduction in hatching success of eggs of the copepod *Centropages hamatus*, and that no eggs survived exposure to 0.08 mg L^{-1} of this crude oil WSF.

3.3.1.3 Behavioural Effects

The narcotizing effect of certain petroleum hydrocarbons is well documented, and many of the behavioural disruptions observed among zooplankton exposed to oil are likely due to this phenomenon. Berdugo et al. (1977) found that hydrocarbon concentrations of 0.2-0.5 mg L^{-1} rapidly caused narcosis in copepods, but recovery was possible as long as exposures were brief (15 minutes or less at oil concentrations of 1.8 mg L^{-1}). Inhibition of zooplankton feeding has been observed in mesoscale experiments and laboratory studies, with several authors reporting that ingestion rates of copepods have been reduced at hydrocarbon concentrations in the range 0.01-1 mg L^{-1} (Berdugo et al. 1977; Berman and Heinle 1980; Cowles and Remillard 1983). Berdugo et al. (1977) observed complete inhibition of feeding at hydrocarbon concentrations exceeding 1 mg L^{-1}. The phototactic responses of zooplankton have also been shown to be disrupted by hydrocarbon exposure. Lee and Nicol (1977) found that a WSF of No. 2 fuel oil disrupted phototaxis at concentrations of 2-4 mg L^{-1}.

These studies clearly indicate that petroleum hydrocarbons can affect the behaviour of copepods, with the threshold oil concentrations required to produce these effects lying in the range from 0.1 to 1 mg L^{-1}. Since this concentration range coincides with the oil levels that have been shown to be acutely toxic (Section 3.3.1.1), it is unlikely that effects of oil on the behaviour of zooplankton would be significant in an actual spill situation. Only effects that occur at concentrations well below those that are lethal (e.g. decreased fecundity and hatching success) may be environmentally significant, and even these would only occur in a small proportion of the total zooplankton in any region affected by a spill.

3.3.1.4 Effects of Oil Spill Countermeasures

The only oil spill countermeasure that could significantly affect zooplankton populations is the use of chemical dispersants, and these may substantially increase the impact of the spilled oil on zooplankton. By increasing the rate of dispersion of oil into the water column, dispersants increase the concentrations of both dissolved and particulate oil fractions in the water column and increase the time that organisms will be exposed to hydrocarbons.

The effects of dispersants and D/O mixtures on zooplankton have been the subject of a number of studies, summarized in two reviews by Wells (1982, 1984). The toxicity of dispersants or their components have been determined for the planktonic life stages of a number of invertebrate phyla including coelenterates (Latiff 1969), annelids (Latiff 1969; Bellan et al. 1971; Akeson 1975); molluscs (Renzoni 1973), copepod and decapod crustaceans (Czyzewska 1976; Doe and Wells 1978; Foy 1979) and echinoderms (Lonning and Hagstrom 1975; Hagstrom and Lonning 1977). These studies have examined the toxicities of dispersants developed in both the pre- and post-Torrey Canyon eras, and have indicated that many of the dispersants developed in recent years are much less toxic to zooplankton than older formulations.

There are substantial differences in the sensitivity of different zooplankton taxa to dispersants. This is undoubtedly related to variability in the formulations of dispersants developed and tested to date, as well as to differences in the physiology of the members of the range of phyla represented in zooplankton communities. For example, Wells (1984) reported that for water-base dispersants, the following order of sensitivity was observed among planktonic life stages: fishes > molluscs > crustaceans. However, the order was reversed when petroleum-base dispersants were used. Wells (1984) also reported that LC_{50} values for a single dispersant (Corexit 9527) and a variety of species ranged from 1 to >1000 mg L^{-1} under different test conditions. Despite this large inter-specific variability in sensitivity, the toxicity of Corexit 9527 was less than that of oil in almost all species tested. Other research has shown that this is true for a number of other recently developed dispersants. It is therefore unlikely that present generation dispersants will contribute significantly to the toxicity of oil-dispersant mixtures.

3.3.2 Bioaccumulation Kinetics and Enviromental Fate of Hydrocarbons

The role of zooplankton in the environmental dynamics of petroleum hydrocarbons has been the subject of some study over the past decade because (1) this community represents a potential entry point for hydrocarbons into marine food webs, and (2) zooplankton may contribute significantly to the metabolic degradation and sedimentation of hydrocarbons in the water column. Investigations of hydrocarbon dynamics in zooplankton have been briefly reviewed by Corner et al. (1976), Bend and James (1978), and Lee (1982), while more detailed discussions of this area of research are provided in Corner (1978) and Connell and Miller (1981). Available information regarding metabolic degradation of hydrocarbons by marine zooplankton and other invertebrates has also been reviewed by Lee (1981).

The concentration of any hydrocarbon in living tissue and the rate of flux of hydrocarbons through a population are both determined by a number of factors including hydrocarbon exposure conditions (i.e. the nature of the compound and whether it is in solution or bound to food particles) and the processes of hydrocarbon uptake, distribution among tissues, metabolic degradation and depuration. The interrelationships among these processes are discussed in the section on hydrocarbon bioaccumulation kinetics in fishes (Section 3.4.1.5).

Little is known about the accumulation of hydrocarbons from water by zooplankton and the few exisiting studies have focused on uptake of naphthalenes. These studies show that naphthalene is rapidly metabolized once it has entered zooplankton tissues. In addition, much of the parent compound and some of its degradation products appear to be very quickly excreted, although a small proportion of the metabolites appear to be retained in tissues for prolonged periods. Similar patterns of hydrocarbon accumulation, metabolism and depuration have been observed in both copepods (Lee 1975) and shrimp larvae (Sanborn and Malins 1977). Research has shown that when zooplankton were exposed to C^{14}-labelled naphthalene in solution for 12-24 h, levels of radioactivity in tissues were 50-100 times higher than in the exposure media. When these organisms were subsequently placed in uncontaminated media, the loss of the C^{14}-label was usually rapid and biphasic. Ninety percent or more of the label in the organisms was quickly lost in what was termed the "fast-clearing phase"; the biological half-life of these hydrocarbons was 2-6 h. The remaining label was lost as a "slow-clearing phase", where the biological half-life of the compounds (likely naphthalene degradation products) was 75 h or more. Using a two-compartment model and clearance rate constants derived from the data of Sanborn and Malins (1977) for larval spot shrimp, a value of approximately 20 g $g^{-1}h^{-1}$ can be obtained for the coefficient of transfer of naphthalene from water to zooplankton. In other words, 1 g of shrimp is capable of removing the naphthalene from 20 g of seawater in 1 h. Harris et al. (1977) demonstrated that uptake rates for naphthalene are directly related to its concentration in water over a range of 1-100 mg L^{-1} (i.e. napthalene uptake is a first-order process over this range of concentrations). These authors also found that naphthalene uptake rates were highly dependent on body size, while Trucco et al. (1983) and Eastmond et al. (1984) reported that, as might be expected, both uptake and excretion rates varied from compound to compound for a range of aromatic hydrocarbons. Harris et al. (1977) also demonstrated that ambient temperature had a major influence on the accumulation of naphthalene by the copepod Calanus finmarchicus; accumulation rates were inversely related to temperature over the range 5-16°C. These authors speculated that this was due to the differential effects of temperature on rates of hydrocarbon uptake and metabolic degradation, with the rate of degradation increasing more rapidly with increasing temperature than the rate of uptake.

Corner et al. (1976) and Harris et al. (1977) examined the accumulation of naphthalene from media containing food particles to determine the potential importance of this pathway. Both groups concluded that hydrocarbon uptake in contaminated prey could be a significant pathway for accumulation in zooplankton. However, McCarthy (1983) demonstrated that accumulation rates of benzopyrene and anthracene were lower from media containing food than from media without prey or other particulates. This suggested that accumulation of these compounds was predominantly from the water. The relative importance of food and water vectors for hydrocarbon uptake by zooplankton therefore appears to depend on (1) the concentration of food and other particulates present in the water, and (2) the specific hydrocarbon and its relative affinity for water, zooplankton and particulate matter.

The metabolic degradation of petroleum hydrocarbons by zooplankton was first examined by Lee (1975), who observed that several groups of zooplankters including copepods, euphausiids, amphipods and crab zoea were capable of metabolizing a range of aromatic hydrocarbons, whereas ctenophores and jellyfish were not. The ability of organisms to metabolize petroleum hydrocarbons appears to depend on the presence in the tissues of the mixed function oxidase (MFO) system. Annelids, arthropods, echinoderms and molluscs have been shown to possess this enzyme system, which appears to function similarly in vertebrates and invertebrates (Lee 1981).

Studies of bioaccumulation kinetics in zooplankton have shown that hydrocarbons are rapidly excreted, usually within 24-48 h after transfer of zooplankton to oil-free media. As a result, it is probable that hydrocarbons accumulated by zooplankton after oil spills would be depurated within a few days, thereby limiting the potential for transfer of these compounds to members of higher trophic levels in marine food webs. On the other hand, there is evidence that substantial biomagnification of hydrocarbons may occur within communities that are exposed to chronic sources of oil pollution. For example, Dillon (1982) observed a 10-fold biomagnification of dimethylnaphalene during a 32-day experiment involving a single-step transfer of hydrocarbons to *Artemia salina*. However, bioaccumulation of petroleum hydrocarbons by marine invertebrates has not yet been unequivocally demonstrated in areas subjected to chronic oil inputs.

The research of Corner et al. (1976) demonstrated that (1) the excretion of accumulated naphthalene by zooplankton was very rapid; and (2) more than 50 percent of the naphthalene taken up was excreted in the form of metabolites. This suggests that zooplankton could play a significant role in the metabolic degradation of hydrocarbons in the water column. The research of Harris et al. (1977) lends support to this hypothesis. These authors observed steady-state concentration factors of 1000 and 50,000 for the copepods *Calanus helgolandicus* and *Eurytemora affinis*, respectively, when these organisms were simultaneously exposed to naphthalene through food and water vectors. Using a conservative estimate of 12 h for the biological half-life for naphthalene excretion, these data yield estimates of uptake rate constants of 60 and 3000 g of water cleared of naphthalene per gram of copepods per hour. (The large difference in the estimated uptake rates for the two species may be due to large differences in body size). However, even with these rapid rates of flux of hydrocarbons through *Calanus* and *Eurytemora*, one gram of copepods would metabolize only 0.006 to 0.3 percent of the naphthalene in one m^3 of seawater per hour. It is therefore considered unlikely that zooplankton could contribute significantly to the metabolic degradation of hydrocarbons in the water column.

Several authors have suggested that filter-feeding zooplankton (particularly copepods) may greatly accelerate the sedimentation of hydrocarbons from the water column through ingestion of oil droplets and food particles containing oil, and subsequent incorporation of this oil into fecal pellets which sink to the sea floor. Earlier research by Conover (1971) showed that copepods do ingest oil particles which subsequently become incorporated in their fecal pellets. In a study of the fate of oil spilled following the grounding of the ARROW, Conover (1971) concluded that perhaps 20 percent of the Bunker C that appeared the water column as particles may have been sedimented in zooplankton fecal pellets. Sleeter and Butler (1982) estimated that 8-23 mg $m^{-2}y^{-1}$ of hydrocarbons were sedimented out of the water column in the Sargasso Sea in fecal pellets. This is equivalent to roughly 30-50 percent of the particulate oil that annually enters the Sargasso Sea. These results suggest that fecal sedimentation may be an important mechanism for the transfer of hydrocarbons from the pelagic zone to the sediments.

3.3.3 Controlled Microcosm Experiments

Controlled microcosm experiments have been conducted to study the effects of oil on organisms under natural conditions, and to examine the effects of oil on interactions among members of the pelagic community. The studies have been undertaken by a number of research teams including Lee and Takahashi (1977), Lee et al. (1978), Davies et al. (1980), Oviatt et al. (1982), and Dahl et al. (1983). The experiments have demonstrated a number of responses by zooplankton to oil exposure including reduced feeding rates and changes in community structure. For the most part, these investigations have confirmed the findings of smaller scale laboratory research. One of the more reproducible and significant observations has been some zooplankton mortality at hydrocarbon concentrations in the range from 0.02 to 0.1 mg L^{-1}, which is substantially lower than acute toxicities determined from laboratory investigations (Section 3.3.1.1).

3.3.4 Experience From Past Oil Spills

Studies of zooplankton have followed several oil spills (Duval et al. 1981). Extensive mortality of zooplankton has only been documented in a few cases such as the AMOCO CADIZ spill (Samain et al. 1979). Sublethal effects of oil on zooplankton may occur following most spills, although they have only been examined or discussed to a limited degree. For example, Johansson et al. (1980) speculated that the apparent increase in phytoplankton biomass observed after the TSESIS spill was attributable to a reduction in grazing rates by zooplankton. Samain et al. (1979) reported a depression in the activity of digestive enzymes in copepods collected near the site of the AMOCO CADIZ spill. However, by far the most commonly reported zooplankton-oil interactions following spills are the ingestion of oil droplets by zooplankton and external contamination of some zooplankton with oil (Conover 1971; Polak et al. 1978; Samain et al. 1979; Johansson et al. 1980).

Studies conducted after spills suggest that while effects of oil on zooplankton can be detected, there are seldom any major changes in biomass. The changes in zooplankton abundance that do occur are both highly localized and short-term.

3.3.5 Vulnerability and Recovery Potential

It is clear that zooplankton populations are sensitive to the effects of hydrocarbons, and that lethal and sublethal effects have occurred following some spills. However, it is highly unlikely that significant damage to zooplankton populations would result from oil spills because any effects would be short-term and occur in localized areas, whereas zooplankton populations are widely distributed. Once elevated hydrocarbon concentrations in the water column have been reduced through both physical and biological processes, local zooplankton populations would be expected to recover rapidly through immigration of individuals from adjacent areas. Even in areas where recruitment by immigration is low, recovery would occur within a year through reproduction.

Arctic zooplankton populations may be somewhat more vulnerable and less resilient to oil spills than zooplankton in temperate waters. This is due to the greater persistence of hydrocarbons at low temperatures and the potential for the presence of an ice cover to retard the spreading and dispersal of surface oil. These factors may be offset to some extent by the fact that there is no wave action to disperse oil into the water column during periods of ice cover, although some dispersion of oil not encapsulated in the ice would still occur as a result of subsurface currents. However, many arctic zooplankters have longer generation times than temperate species, which may reduce recovery potential to some extent.

SECTION 3.4

FISH

Ken Trudel

The following sections summarize available information on the fate and effects of petroleum hydrocarbons on finfish. The review includes both pelagic and demersal species and their juvenile stages (eggs and larvae). Demersal species or stages are more likely to be affected by oil in benthic habitats than by hydrocarbons in the water column because of the greater persistence of oil in sediments than in the water column. Pelagic fish eggs and larvae may also be exposed to hydrocarbons in a variety of habitats, but vulnerability is greatest for species whose eggs and larvae occur at or near the water surface where hydrocarbon concentrations are highest following spills.

The effects of petroleum hydrocarbons on pelagic and marine fishes are of ecological and economic concern. Fishes are important predators and prey in marine ecosystems and form the basis of major commercial fisheries. In the Canadian Arctic, some species are also harvested by coastal communities. Acute and chronic oil discharges may affect fish populations through the toxic and sublethal effects on eggs, larvae, juveniles and adults. Coastal migrations of anadromous species may also be disrupted as a result of acute or chronic exposure to petroleum hydrocarbons.

The concentration and persistence of dispersed and dissolved oil in the water column can vary substantially between nearshore and offshore spills. Oil spills are point source discharges that are usually of short duration. Consequently, in offshore regions, hydrocarbon concentrations in the water column are only elevated over a limited area and for a short period (a few hours to a few days). On the other hand, fish populations may be subjected to more prolonged exposure when spills occur in confined inshore waters with limited mixing and low exchange rates. Chronic petroleum discharges from ships and shore-based installations also may affect fish populations to a greater degree in nearshore and inshore habitats. In addition, there is a greater potential for contamination of sediments by hydrocarbons in shallow inshore areas, thereby creating a source of chronic oil release to adjacent waters.

The effects of oil spills and chronic oil pollution on fishes have been the subject of a large number of laboratory and field studies. The results of this research have also been summarized by numerous authors (Moore and Dwyer 1974; Corner et al. 1976; Rice et al. 1977; Whipple et al. 1978; Connell and Miller 1980; Malins and Hodgins 1981; Rice 1981; McIntyre 1982; Penrose 1982; Rice 1985). The effects of past spills on fish populations were also reviewed by Duval et al. (1981). Other authors have discussed the implications of oil spills for arctic fish populations (Johnson 1983; Clark and Finley 1984; Nelson-Smith 1984; Rice 1984).

This review focuses on the effects of oil spills considered most relevant to cold Canadian marine environments. The effects of temperature on sensitivity of arctic and temperate fishes to hydrocarbons and on hydrocarbon bioaccumulation are considered. As indicated earlier, information on the lethal and sublethal effects of oil on all life history stages of pelagic and demersal fishes is reviewed. Separate sections also address the effects of oil spill countermeasures on fisn populations, as well the bioaccumulation of hydrocarbons and tainting of fishes.

3.4.1 Sensitivity of Fish to Oil and Petroleum Hydrocarbons

3.4.1.1 Lethal Effects of Petroleum Hydrocarbons on Adult Fish

Mortality of adults has occurred as a result of oil spills, generally following spills in relatively confined areas such as bays or estuaries, or when adult fishes have been unable to avoid oil (e.g. fish in mariculture pens; Duval et al. 1981). For example, mortality of adult fishes was reported following the West Falmouth spill (Hampson and Sanders 1969), the JULIANA wreck in Japan (Motohiro and Inoue 1973) and the AMOCO CADIZ spill (Maurin 1981). Other spills that have resulted in confirmed reports of fish mortality are described in Duval et al. (1981). Therefore, under certain circumstances, fishes may be exposed to lethal concentrations of oil. This section describes the exposure conditions necessary for fish mortality, and the influence of environmental variables on the sensitivity of fish to hydrocarbons.

Several reports suggest that the age of the fish is the single most important factor affecting its sensitivity to hydrocarbons. In 96-h bioassays involving exposure to benzene, Moles et al. (1979) demonstrated that the sensitivity of pink salmon increased by almost two orders of magnitude during development from egg (96-h LC_{50}: 340 $\mu L\ L^{-1}$) to fry (96-h LC_{50}: 5 $\mu L\ L^{-1}$). Similar findings have been obtained for other species (Moles 1979), for other compounds (Sharp et al. 1979), and crude oils (Rice et al. 1975).

By comparison with the influence of age, interspecific variation in sensitivity to a given compound appears to be relatively small. Rice et al. (1979) conducted bioassays on several fish species exposed to benzene, and reported that the 96-h LC_{50} value for the most resistant species was only twice that determined for the most sensitive species. However, more than one order of magnitude of variation in sensitivity to Cook Inlet crude oil was observed among a range of Alaskan fish and invertebrate species tested.

There are substantial differences in the toxicity of different crude oils and refined petroleum products to marine fishes, because both crudes and refined products are complex mixtures of hydrocarbons that differ in their physical and chemical properties. Anderson et al. (1974) and Moore and Dwyer (1974) concluded that toxicities of the water-soluble fractions (WSFs) of an oil were due largely to the high proportion and persistence of naphthalenes in the WSF, and the relatively high toxicity of naphthalenes. For this reason, some authors have expressed oil concentrations in terms of total aromatics or total naphthalenes (Rice et al. 1977; Korn et al. 1979). This approach appears to reduce variation in the apparent toxicity of different oils. Based on studies with WSFs, 96-h LC_{50} values for the toxicity of petroleum hydrocarbons to small fishes range from 1 to 20 mg L^{-1} (Anderson et al. 1974; Rice et al. 1977; Moles et al. 1979).

Physiological and environmental variables such as salinity, food abundance and stress (e.g. parasitism) affect the sensitivity of fishes to petroleum hydrocarbons (Rice 1985). However, the influence of temperature on the toxicity of oil to fish is of greatest interest in this review. Temperature may be expected to influence the toxicity of hydrocarbons by affecting: (1) their persistence in the exposure medium; (2) the metabolic rate of fishes and hence rates of hydrocarbon uptake, metabolism and depuration; and (3) the intrinsic toxicity of specific hydrocarbons to fishes. However, Korn et al. (1979) found that the sensitivity of pink and chinook salmon fry to naphthalene, toluene and a WSF of Cook Inlet crude oil was only slightly affected by changing temperatures over the range from 4° to 12°C.

3.4.1.2 Sublethal Effects of Petroleum Hydrocarbons on Fishes

Oil exposure has been shown to cause a range of sublethal physiological, pathological and behavioural effects in fishes, although most of these effects have only been observed in laboratory experiments rather than following actual oil spills. Many of the sublethal effects observed in fishes exposed to petroleum hydrocarbons are minor, short-term and completely reversible. Some changes may be adaptive such as increased respiration rate (Rice et al. 1977; Thomas and Rice 1979) or elevated serum glucose levels associated with alarm or stress reactions (Dimichele and Taylor 1978; Thomas et al. 1980). Other physiological symptoms reflect pathological changes such as the elevated serum electrolyte levels associated with disrupted osmo-regulation (Chapman et al.; Thomas et al. 1980). These changes are of very short duration and disappear very quickly following cessation of exposure. Other sublethal effects such as disruption of growth or energy metabolism (Moles and Rice 1983) may be of longer term significance. A number of histopathological conditions have been observed in field studies or laboratory experiments. Although it is difficult to interpret these conditions in ecological terms, they may be useful indicators of recent exposure to petroleum hydrocarbon pollution. Some of the pathological changes and abnormalities observed in fishes include: (1) liver necrosis (Sabo and Stegeman 1977; Dimichele and Taylor 1978; McCain et al. 1978; Woodward et al. 1983); (2) gill damage (Blanton and Robinson 1974; Ernst et al. 1977; Dimichele and Taylor 1978; Englehardt et al. 1981; Woodward et al. 1983); (3) necrosis of intestinal mucosa (Dimichele and Taylor 1978; Hawkes et al. 1980); (4) pancreatic necrosis (Dimichele and Taylor 1978; Solangi and Overstreet 1982); (5) renal damage (Dimichele and Taylor 1978); (6) necrosis of olfactory and taste organs (Gardner 1975; Solangi and Overstreet 1982); (7) muscle degeneration (Dimichele and Taylor 1978); and (8) changes in fin pathology (Woodward et al. 1983). Some of these histopathological changes are unique to hydrocarbon exposure, and therefore may be useful in monitoring programs designed to detect chronic oil pollution. However, other pathological effects appear to be caused by stress and therefore could be a reflection of environmental conditions unrelated to hydrocarbon exposure.

3.4.1.3 Sensitivity of Fish Eggs and Larvae

It is important to consider the effects of oil on fish eggs and larvae separately from effects on older life stages because: (1) the eggs and larvae of some species are more sensitive to oil than adult stages (Rice et al. 1975); (2) some of these stages are also more vulnerable to oil exposure because they are present in the upper portion of the water column (larvae) or on the water surface (eggs of some species) where oil concentrations are highest following spills; and (3) while many adult or juvenile fish have been shown to avoid oil-contaminated waters, planktonic eggs and larvae are unable to avoid oil exposure.

A number of researchers have examined the effects of petroleum hydrocarbons on fish eggs and larvae in either field or laboratory studies. The effects of a limited number of oil spills on fish eggs have also been investigated (Duval et al. 1981). The results of these research programs are summarized in Table 3.4-1. Effects range in severity from minor ones such as reduced size of larvae at hatching to more severe effects such as embroynic death. Some effects (e.g. egg death) result in the immediate loss of individuals from the population, while others (e.g. reduced larval size) reduce the probability that the individual(s) will survive to adulthood. Some sublethal effects are reversible and should not affect the long-term survival of individuals within the population. From an ecological perspective, the most important consideration is the relationship between hydrocarbon exposure and responses such as embryonic or larval death or gross larval deformity, all of which affect the proportion of viable larvae within a population. A first-order approximation of the sensitivity of larvae to hydrocarbons is shown in Figure 3.4-1, which summarizes recent data on larval mortality as a function of oil concentration. The data indicate that hydrocarbon concentrations less than 0.6 mg L^{-1} do not adversely affect larval survival. It should be emphasized that since these results are based on experiments where individuals were continuously exposed to hydrocarbons from the time of fertilization to hatching, they should not be applied to spills in offshore waters where hydrocarbon exposure would be relatively brief (e.g. 12-48 h).

There is only limited information on the sensitivity of fish eggs and larvae to brief hydrocarbon exposure. Stoss and Haines (1979) reported that eggs of the Japanese medaka were only slightly more sensitive to a 96-h exposure to toluene than to a 24-h exposure. Tilseth et al. (1984) documented irreversible inhibition of feeding in cod larvae exposed for 2 h to 0.9 mg L^{-1} (WSF) of Ekofisk crude oil. However, the larvae were unaffected by 1-h exposure to 0.6 mg L^{-1} of the Ekofisk crude. These results suggest that even very brief exposures of fish larvae to hydrocarbons at concentrations from 0.6-6.0 mg L^{-1} may cause adverse effects comparable to those observed during prolonged exposures.

TABLE 3.4.1 Summary of deleterious effects of oil and petroleum hydrocarbons on young stages of fishes

Response	Reference
Embryonic malformation	1
Death of embryo	2,4,5,7
Failure to hatch	1,3,7
Mortality of larvae	1,3
Morphological aberrations in larvae	1,3,5,7
Physiological effects - loss of buoyancy	5
Behavioural abnormalities in larvae	
- reduced feeding rate	3,5,6
- reduced swimming ability	5,6
Reduced size at hatching (length, weight)	1,3,5,6

1. Linden (1978)
2. Moles et al. (1979)
3. Struhsaker et al. (1974)
4. Stoss and Haines (1979)
5. Rice et al. (1975)
6. Tilseth et al. (1984)
7. Smith and Cameron (1979)

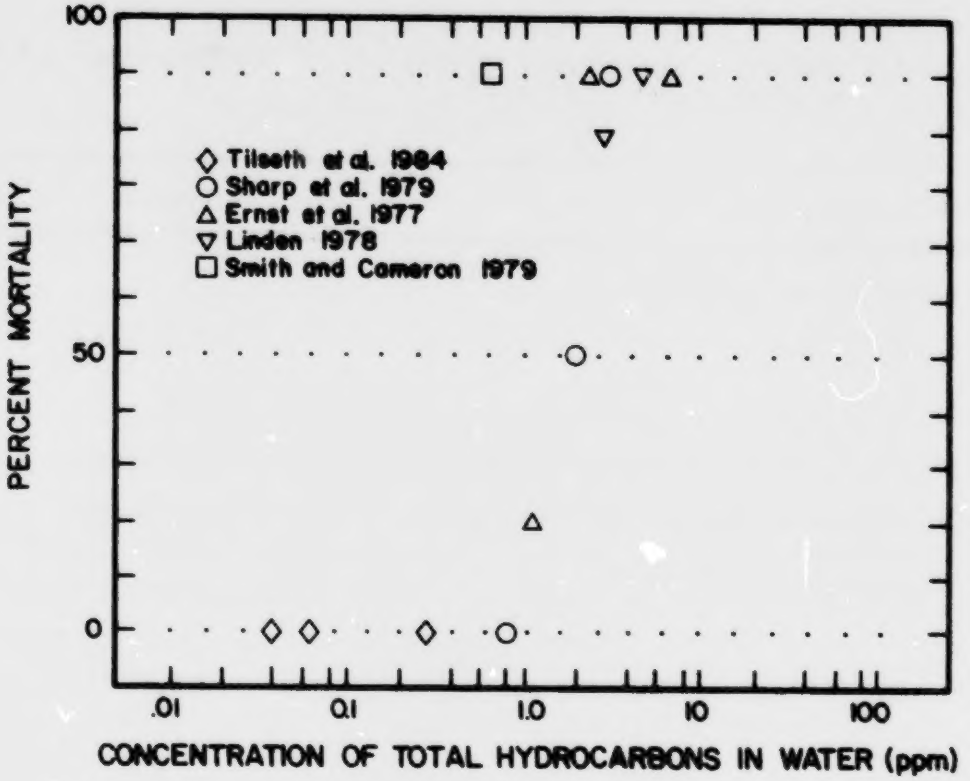

Figure 3.4-1 Sensitivity of the Embryo-Larval Stages of Fish to Water-Soluble Fractions of Crude and Refined Products Under Conditions of Continuous Exposure

The influence of temperature on the sensitivity of fish eggs and larvae to petroleum hydrocarbons has not been well documented. Linden (1978) reported that over a temperature range of 9°C to 14°C, lower temperatures slightly reduced the sensitivity of juvenile herring to the WSFs of light fuel oils and crude oils.

3.4.1.4 Effects of Petroleum Hydrocarbons on Fish Behaviour

There are at least two significant issues related to the effects of petroleum hydrocarbons on the behaviour of fishes. The first and most general question is whether fishes are capable of detecting and avoiding elevated hydrocarbon concentrations in the water that result from spills. The second question, which is more applicable to northern climates, is whether the annual spawning migrations of anadromous fish species are likely to be disrupted or delayed by the presence of hydrocarbons in the water column.

It is commonly believed that adult pelagic fish can detect and thereby avoid high concentrations of hydrocarbons in the water column, although information in support of this hypothesis is somewhat limited and circumstantial in nature. Various investigators have documented active avoidance responses of fish to high concentrations of hydrocarbons in laboratory bioassay and sublethal research programs. For example, during studies of the sublethal effects of Prudhoe Bay crude oil (WSF) on juvenile coho salmon, Duval and Fink (1979) noted strong avoidance responses by fish in flow-through exposure vessels as soon as hydrocarbon concentrations reached 3.2 mg L^{-1}. These responses were not observed in coho exposed to lower hydrocarbon concentrations or held in control vessels. Although the results of this and other studies (see review by Olla 1981) suggest that pelagic fish may avoid waters containing high concentrations of hydrocarbons, the evidence is not conclusive. Sprague and Dury (1969) exposed rainbow trout to phenol and other toxicants, and reported that while this species avoided other contaminants (e.g. detergents, Kraft pulp mill effluents), they did not avoid elevated phenol concentrations (even concentrations that were lethal). In a similar manner, Westlake et al. (1983) found that rainbow trout did not exhibit avoidance responses in water containing a steep gradient in concentrations of petroleum refinery effluent. However, Hellstrom and Doving (1983) described a variety of avoidance responses by cod during exposures to oil. The types of responses that were observed during exposure to the WSF of diesel oil included increased activity, spontaneous darting, and swimming backward as fish descended to the bottom. Each of these responses would be effective in moving fish out of localized areas containing high hydrocarbon concentrations. Hellstrom and Doving (1983) also reported that some individuals displayed avoidance behaviour to oil concentrations as low as 1 ppb, although strong responses were only consistently observed to concentrations of 1 mg L^{-1} or greater. The results of this research indicate that cod display avoidance behaviour at hydrocarbon concentrations which are near the lower limit of those that have been shown to be acutely toxic, and may therefore be able to avoid areas containing elevated hydrocarbon concentrations.

Anadromous fish are important to commercial, recreational and domestic fisheries in many Canadian coastal areas. Oil spills in nearshore environments, estuaries and rivers could disrupt or delay spawning migrations of these species to natal freshwater spawning habitats. Although there has been no documentation of this, or of reduced harvest of anadromous fish due to oil spills, Sprague et al. (1965) reported that the annual spawning migration of Atlantic salmon in the Miramichi River was affected by copper and zinc pollution. Atlantic salmon encountering elevated copper and zinc concentrations aborted their migrations and moved downstream without spawning.

The migratory behaviour of salmonids could be disrupted by the presence of petroleum hydrocarbons in the water column. The research of Rice (1973) and Maynard and Weber (1981) with juvenile pink and coho salmon indicates that downstream migrants avoid areas containing lethal concentrations of hydrocarbons. However, the impact of oil on the population of migrant fish in this situation would likely be small because these oil concentrations would persist for only a short period relative to the time frame over which downstream migration occurs. Weber et al. (1981) examined whether the "homing" instinct in adult coho salmon would override any short-term response to petroleum hydrocarbons. One of two parallel fish ladders was contaminated with a mixture of monocyclic aromatic hydrocarbons, and upstream migrants avoided this portion of the facility when hydrocarbon concentrations were approximately 1 mg L^{-1} or greater. These authors reported that a hydrocarbon concentration of 5 mg L^{-1} virtually prevented upstream migration through the contaminated side of the fishway, although migrant coho did not abort their migration and simply used the alternate ladder. This study clearly suggests that the homing behaviour and spawning migration of salmon may be at least temporarily disrupted by oil spills in migration corridors.

3.4.1.5 Tainting and Bioaccumulation Kinetics

There is substantial evidence that fish accumulate petroleum hydrocarbons from water, sediments and their prey organisms. Although these compounds may not reach tissue concentrations high enough to be lethal or to cause histopathological changes, they are still of concern to ecologists and fisheries biologists. Low-level hydrocarbon contamination is a problem for the fisheries since certain hydrocarbons or their degradation products may cause tainting. In addition, some compounds are potentially carcinogenic, and if passed along food chains from fish to their predators and man, may constitute a health hazard.

Tainting is a change in the characteristic smell or flavour of fishes (or other harvested resources), and is of concern because it may decrease desirability and marketability of the catch. The presence of "kerosene like" or "oily" flavour has been clearly correlated with the presence of petroleum hydrocarbons in fish flesh from areas where spills of crude oil or refined petroleum products have occurred (Mackie et al. 1972; Paradis and Ackman 1975), or where there have been chronic discharges from ships or shore-based industrial facilities (Nitta 1970; Vale et al. 1972). Ironically, chronic oil pollution resulting in chronic tainting of fishes has created an accepted unique flavour in some species important in Japan (J. Payne, pers. comm.), and here removal of the source of tainting would very likely decrease the marketability of certain species.

Tainting is caused by external contamination of fishes, or by uptake of hydrocarbons from the water, sediments or prey organisms and accumulation of these compounds or their derivatives in fish tissues. In general, oil-contaminated sediments represent the most serious area of concern with respect to tainting because hydrocarbons may persist for years in sediments and therefore represent a source of chronic hydrocarbon exposure. In contrast, elevated hydrocarbon concentrations in the water column are generally short-term, except in areas proximate to chronic oil inputs. The "oily" flavour or odour associated with tainting can be caused by a wide variety of hydrocarbons including alkanes, aromatics, phenolics and other sulfur-containing compounds. The organoleptic response to a compound is a function of two factors (1) the amount of the compound present in fish tissues, and (2) the strength of the flavour of that compound. Tainting is considered to exist when the concentration of any compound in fish tissues exceeds the taste threshold for that compound. Many of the compounds that have been associated with fish tainting and their taste thresholds are listed in Table 3.4-2.

As in all metazoans, the accumulation of hydrocarbons in fishes is the net result of four pharmacokinetic processes: uptake, distribution (among the various tissues), metabolic degradation and depuration. Fishes can take up hydrocarbons directly from the water, by assimilation from food, and in some species, by complex transfer processes from sediments. The hydrocarbons then become distributed among the various tissues according to the relative properties of each compound and tissue. In some tissues, hydrocarbons are metabolized by a range of enzyme systems. The rate of conversion is a function of the nature of the compound and the specific tissue where it resides. Eventually both the parent compound and its metabolites are excreted by various routes, at rates that are determined largely by the nature of the compounds.

TABLE 3.4-2 Summary of available information on tainting petroleum hydrocarbons

Compound	Taste Threshold, ppm
n-hexane	1[a]
cyclohexane	5[a]
hexene	1[a]
benzene	0.5[b]
toluene	0.25[a]
xylene	0.3[b]
ethyl benzene	0.25[c]
phenols	1[c]
napthlalene	1[c]
mercaptans	0.06-0.2[d]

a. Ogata and Miyake (1973)
b. OHM-TADS (1981)
c. Connell and Miller (1981)
d. Connell and Miller (1981); compounds such as N-butyl mercaptan and ethanethiol.

The relationships among some of these processes can be illustrated by a simple model which shows the exchanges of a compound between an organism and its aqueous medium (Figure 3.4-2). The processes depicted are: (1) uptake of hydrocarbons from the water; (2) their metabolic degradation; and (3) excretion of both the parent compounds and their metabolites. Figure 3.4-3 shows data from Southworth (1979) on the accumulation of anthracene by fathead minnows (Pimephales promelas) during a 250-h period of continuous exposure. Within 50 h of exposure to 0.25 ppb anthracene, the concentration of anthracene within the fish reached a steady-state equilibrium with the water. At steady-state, the concentration factor (ratio of the concentration in tissue over that in water) was 400-500, and the rate of uptake was equal to the combined rates of excretion of the parent compound and its metabolic degradation products. Both the time sequence of pollutant uptake and steady-state concentration factor observed for anthracene by Southworth (1979) are consistent with the kinetic rate constants determined independently by Spacie et al. (1983).

The latter authors evaluated the rate constants for each of the following processes with anthracene:

Uptake Rate: 36 g water cleared of anthracene g^{-1} fish tissue h^{-1}
Fractional Loss Rate (Parent Compound): 0.04 h^{-1}
Fractional Rate of Metabolic Degradation
of Anthracene: 0.02-0.04 h^{-1}

The uptake, transformation and depuration of hydrocarbons are influenced by a number of variables including the properties of the compound, the target species, its body size and nutritional status, and ambient temperature and water quality. The following sections briefly describe each bioaccumulation process and some of the factors that are known to influence hydrocarbon uptake, metabolism and depuration.

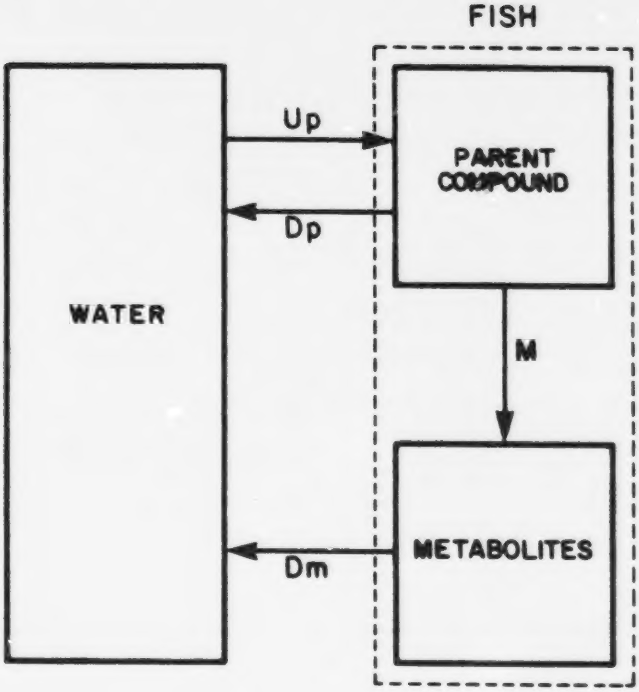

Figure 3.4-2 Simple Model Describing Certain Bioaccumulation Processes Involved in Accumulation of Hydrocarbons From Water by Fishes. Up = Uptake of Parent Compound; Dp, Dm = Depuration of Parent Compound and Metabolites, respiration; M = Metabolic Degradation

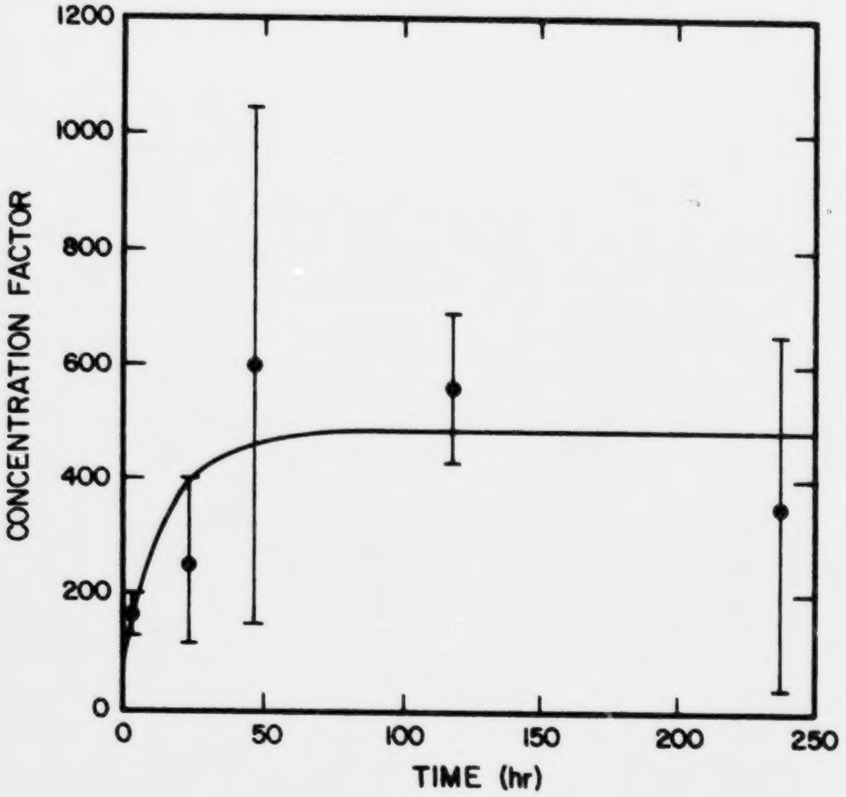

Figure 3.4-3 Bioaccumulation of Anthracene from Water by the Fathead-Minnow (From Southworth 1979)

a) Hydrocarbon Uptake

The uptake of hydrocarbons directly from the water is primarily by direct transfer across the gills. Although there is little direct evidence of this phenomenon, Norstrom et al. (1976) demonstrated a close correlation between rates of uptake of a variety of pollutants and uptake of oxygen in fishes. This strongly suggests that the gills are the dominant route of uptake rather than other possible routes of entry such as passive absorption through the skin. Since the efficiency of hydrocarbon exchange from water to fish is independent of the ambient hydrocarbon concentration, within certain limits, the rate of uptake increases in direct proportion to increasing hydrocarbon concentrations in the water (Spacie et al. 1983).

The rate of uptake of hydrocarbons from contaminated prey is a function of: (1) rate of ingestion of food; (2) hydrocarbon concentrations in prey organisms; and (3) assimilation efficiencies of specific hydrocarbons present in the food. This hydrocarbon uptake vector may be significant among benthic or demersal species, but is unlikely to be a dominant pathway of hydrocarbon accumulation in offshore pelagic species. In recent years, several researchers have investigated the bioaccumulation of ingested hydrocarbons, although there has been no attempt to assess the assimilation efficiency of hydrocarbons from ingested prey.

The transfer of hydrocarbons from sediments to fishes occurs through two pathways (1) direct transfer from the sediments to the skin of a demersal fish, and (2) desorption of hydrocarbons from the sediments to interstitial waters, followed by uptake from water via the gills. No research so far has elucidated the actual uptake mechanism(s), although the results of some studies do allow estimation of the concentration factors from some compounds. For example, concentration factors of parent hydrocarbons have been determined for liver and muscle tissues of the English sole (Table 3.4-3). These values should not be considered steady-state concentration factors since the hydrocarbon levels in the sediments declined significantly in some cases during the course of the experiments. Nevertheless, these data suggest that concentration factors vary among hydrocarbons. Values for naphthalene compounds ranged from 0.5 to 15, whereas those for benzopyrene were much lower at 0.001-0.004 (McCain et al. 1978; Varanasi and Gmur 1981; Stein et al. 1984).

b) Metabolic Transformation

Many petroleum hydrocarbons are altered through metabolic processes after they enter fish tissues. Although there is a variety of possible metabolic pathways, several authors have proposed a general two-step mechanism. The first is an oxidation step that is mediated by a group of enzymes called mixed-function oxidases (MFOs) and results in an oxygenated

TABLE 3.4-3 Summary of observed concentration factors for naphthalenes and benzopyrene between sediments and a groundfish, the English sole, based on muscle and liver concentrations of the parent compounds.

Compounds	Muscle	Liver
Naphthalene	3[a] 1.2[b]	1.3[a] 8-15[b]
1-Methylnaphthalene	1[a]	2.5[a]
2-Methylnaphthalene	0.5[a]	3.5[a]
Benzopyrene	0.001[c] 0.02[b]	0.001[c] 0.004[b]

a. McCain et al. (1978)
b. Varanasi and Gmur (1981)
c. Stein et al. (1984)

product. This is followed by a conjunction step where these oxides are enzymatically conjugated with a substrate such as glutathione (Connell and Miller 1981). The net result of these reactions is to alter the toxicity and lipophylicity of the hydrocarbons. Metabolic degradation processes are generally considered to be detoxifying mechanisms because they reduce the affinity of hydrocarbon molecules for membranes. However, the oxidation of aromatics and certain other hydrocarbons results in the formation of several arene or alkene oxides that have a high affinity for DNA or RNA; some of these compounds may be carcinogens or mutagens (Connell and Miller 1981).

The rates of metabolic degradation of petroleum hydrocarbons are highly variable and are influenced by a number of factors. Spacie et al. (1983) demonstrated that degradation rates vary from compound to compound since *in vitro* transformation rates of anthracene and benzopyrene in bluegill sunfish, not previously exposed to hydrocarbons, were 2 and 36 percent of the body burden per hour, respectively. Much of this metabolic degradation activity occurs in the liver but some activity has been observed in other tissues as well. In any given tissue, rates of degradation appear to be a function of the concentrations of both substrate and the enzyme(s) (Schnell et al. 1980; Spacie et al. 1983). However, the rates of transformation are somewhat higher in individuals previously exposed to hydrocarbons (Nava and Englehardt 1982).

c) Depuration

The principal route of depuration for many aromatic hydrocarbons and their metabolites is through biliary excretion (Lee et al. 1972). However, Thomas and Rice (1981) found that the gills were a major pathway for excretion of low molecular weight aromatics, while Lee et al. (1972) reported that the urinary system was an important pathway for excretion of water-soluble metabolites. The rates of excretion of either parent hydrocarbons or their metabolites are poorly documented, largely because rate measurement is complicated by the simultaneous interconversion of the parent compounds and metabolites. However, it is expected that depuration of petroleum hydrocarbons is a first-order process, as it is for other materials including metals, organometallics and halogenated hydrocarbons.

d) Effects of Selected Variables on Bioaccumulation Kinetics

A number of factors influence the bioaccumulation of petroleum hydrocarbons including ambient temperature, lipid content of the prey, suspended solids in the aqueous medium, and in particular, the physical and chemical properties of specific hydrocarbons. The lipophylicity of the compound is perhaps the single most important variable. The rate of uptake from water and the rate of depuration of toxic compounds are positively and negatively correlated with lipophylicity, respectively (Neely 1979). There is also some evidence that even the rates of biotransformation of different aromatic compounds may be positively correlated with lipophylicity (Schnell et al. 1980; Spacie et al. 1983), although cause-effect relationships for this have not been examined.

McCain et al. (1978) found that the concentration factors between sediments and fishes varied widely among compounds, and attributed this variation to differences in the bioavailability of compounds from sediments and their depuration rates from fishes.

The marked influence of temperature on bioaccumulation of hydrocarbons by demersal fishes is of particular relevance to this review. Varanasi et al. (1981) found that starry flounder accumulated more than ten-fold higher concentrations of naphthalene in its musculature at 4°C than at 12°C when ingesting naphthalene-contaminated food. These authors concluded that temperature affected both absorption of naphthalene from the gut and its depuration from tissues. This finding suggests that cold-water species might be expected to accumulate hydrocarbons to a greater extent than warm-water species.

3.4.1.6 Oil spill Countermeasures

In some parts of the world, chemical dispersants have been used to clean oiled shorelines and disperse oil slicks at sea since the late 1950's and early 1960's. Issues relating to their effects on fish and other aquatic life have been the subject of several reviews in recent years (Nelson-Smith 1972; Doe and Wells 1978; Dye and Frydenborg 1980; Wells and Harris 1980; Sprague et al. 1982). One of the primary issues discussed by these authors is the potential for dispersants to increase the effects of oil spills on fish populations. There is little doubt that use of dispersants will increase the vulnerability of organisms in the water column, simply because they increase the amount of oil dispersed and dissolved within it. In addition, the toxicity of dispersants themselves may contribute to the impacts of a spill on some organisms. The earliest dispersants, including those used following the TORREY CANYON spill, were as toxic or more toxic than the oils they were used to treat. Farn (1976) reported that 96-h LC_{50} values for these 'first generation' dispersants ranged from 10 to 100 mg L^{-1}. A wide variety of new dispersants has been developed over the last two decades. The Oil Spill Intelligence Report (1984) lists more than 100 dispersants that have been approved for use in various countries, including some "low toxicity" formulations with 96-h LC_{50} values of 1000-10,000 mg L^{-1} (Doe and Wells 1978). In many of the latter products, the dispersant itself contributes little to the toxicity of the oil-dispersant mixture (Doe and Wells 1978; Wells 1984). Consequently, any increase in the impact of spills on organisms within the water column resulting from the use of such dispersants will be associated with increased concentrations of dissolved and dispersed oil rather than with the toxic effects of the dispersant.

In cold marine environments, temperature may affect dispersant-fish interactions in two ways. First, lower temperatures may affect the sensitivity of fishes to dispersants or oil-dispersant mixtures. Secondly, dispersants are less efficient at lower temperatures, and therefore, either larger quantities of dispersant or more effective and potentially toxic formulations may be required to achieve dispersion of surface slicks. There is little available information on the influence of temperature on the toxicity of current "low-toxicity" dispersants. Wilson (1977) examined the effects of temperature on the toxicity of dispersants to fish larvae. While temperature had little effect on the concentration of dispersants causing mortality during prolonged exposure, it did alter the sensitivity of eggs and larvae during short exposures to higher dispersant concentrations. Over the range of 6-15°C, dispersant toxicity decreased at lower temperatures.

A number of authors have demonstrated that dispersants are generally less effective at lower temperatures (e.g. Byford 1982; Lehtinen and Vesala 1984), so that in order to successfully use dispersants in colder climates, higher doses or more effective formulations are necessary. Cost factors are expected to favour the latter alternative. However, among dispersants developed to date, increases in effectiveness have only been achieved with a corresponding increase in toxicity (Tokuda and Arasaki 1977; Bratak et al. 1982).

Due to the effect of dispersants on interfaces, there is some concern that dispersants may influence the uptake of hydrocarbons from the water column, and thereby increase the likelihood of fish tainting. McKeown (1981) reported a small but significant increase in the accumulation of a mixture of organic compounds by juvenile coho salmon when the dispersant BP1100X was present in the test media. However, it is expected that very small increases in the uptake of compounds from the water, such as those observed in this investigation, are unlikely to significantly increase the risk of tainting.

3.4.2 Vulnerability of Fish to Oil Spills

It is important not to simply equate the sensitivity of fishes or other biota to effects of oil spills with the vulnerability of populations to impacts associated with these events. The potential for adverse impacts of a spill on fish depends on: (1) the vulnerability of individuals within the population; and (2) whether significant numbers of individuals will encounter oil; as well as (3) the sensitivity of individuals to the effects of oil.

Due to the hydrophobic nature of petroleum hydrocarbons, they tend to accumulate at interfaces rather than throughout the water column. Elevated concentrations of hydrocarbons can occur within the water column, but these tend to be very transitory, particularly in offshore areas. The highest hydrocarbon concentrations are found at or near the air-water and sediment-water interfaces, and therefore the species and life history stages most vulnerable to effects from spills are those that occur at these interfaces. These include the eggs and larvae of fishes such as Atlantic cod (Gadus morhua) which are present at or near the sea surface for several weeks, and the juveniles or adults of groundfish species that feed on benthic fauna and rest on or near the seafloor. However, among the latter species, only those populations inhabiting nearshore areas can be considered vulnerable since the likelihood of significant contamination of sediments in deeper offshore waters is low.

The vulnerability of some fish populations to oil spills varies with the degree of congregation of spawning adults or eggs and larvae. For example, the eggs and larvae of many species that spawn in offshore areas are not congregated, but rather dispersed over extensive (tens or hundreds of thousands of km^2) areas of ocean surface (e.g. the Southern Grand Bank stock of Atlantic cod). Consequently, even a major spill should affect only a very small proportion of the eggs and larvae present in surface waters. On the other hand, populations of some species such as Atlantic and Pacific herring congregate during reproduction, and deposit spawn within much smaller areas. Such species would be far more vulnerable to the effects of oil spills.

Anadromous fishes such as salmonids and whitefish may also be vulnerable to oil spills since their spawning migrations may be disrupted or delayed by the presence of petroleum hydrocarbons in nearshore waters and estuaries. However, it is very unlikely that entire runs would be affected. Migration to natal streams, and subsequent spawning, occurs over a period of several weeks to months in most anadromous species, and oil spills would only significantly affect surface waters for a matter of days or a couple of weeks.

3.4.3 Recovery of Affected Populations

Some fish populations may be both sensitive and vulnerable to effects of oil spills. The two primary areas of concern are (1) uptake of hydrocarbons leading to tainting, and (2) changes in the abundance of populations due to acute and chronic lethal toxicity of petroleum hydrocarbons. The present section discusses the factors that would influence recovery of fish populations from either tainting or population declines. However, it is emphasized from the outset that the vast majority of oil spills have not resulted in apparent impacts on fish populations (Duval et al. 1981).

The period of time required for reversal of tainting by hydrocarbons should differ significantly between pelagic and demersal fish species. For mobile demersal species or pelagic species, the depuration of petroleum hydrocarbons from fishes is relatively rapid; biological half-lives range from a few hours to a few days depending on the specific compound. Consequently, once hydrocarbon concentrations in the water or sediments decrease to background levels, or the fishes are no longer exposed to these hydrocarbons, depuration processes should rapidly reduce the tissue concentrations of hydrocarbons. Hence, the compounds or metabolites that contribute to tainting should decrease below their organoleptic detection thresholds within a few days or weeks. Pelagic species and mobile demersal species can therefore be expected to recover quickly from tainting. However, oil contaminated sediments may remain elevated for many months or years, and less mobile demersal species that remain in contact with these contaminated sediments may themselves retain hydrocarbons in their tissues for long periods.

In the event that a fish population is significantly affected by an oil spill, its rate of recovery would depend on rates of recruitment of individuals from adjacent areas and the reproductive potential of the remaining individuals in the population. Movement of individuals between fish populations, though known to occur, is poorly understood and even less documented in arctic populations. Hence, the possibility of recovery of fish populations through immigration is not discussed further in this review. As a result, the following discussion is limited to a review of recovery of fish populations through reproduction. The recruitment potential of fishes is discussed in relation to that of other groups of marine organisms.

When a population is reduced by any cause, its recovery is a function of a number of factors including fecundity, maturation rate and longevity. Percy (1982) used the ecological concepts of r- and k-stategies (overall adaptive strategies) in his recent discussion of the recovery potential of arctic species. The k-stategy is the strategy for maximum competitive ability in a stable and saturated environment, and is characterized by long generation times, low fecundity, and a high degree of survivorship of young. The r-strategy, on the other hand, is a strategy for a maximum rate of population increase, and confers on a population the ability to take advantage of periods of favourable conditions in a highly variable and unpredictable environment. This strategy is characterized by shorter generation times, high fecundity, and low survivorship among young.

It can be argued that among marine species, marine birds have adopted a k-strategy displaying longevity, long maturation time, and low reproductive potential but a very high level of survivorship of young. On the other hand, zooplankton follow an r-strategy with short life expectancy, short maturation time and higher rates of fecundity. In general, fish appear to display an r-strategy because, although many species have a high longevity and are slow to mature, their habit of producing vast numbers of eggs (thousands or even millions of eggs per female depending on the species) means that a population can take advantage of highly favourable conditions to produce large year classes. Therefore, even in the unlikely event of a severe reduction in population size caused by an oil spill, most fish populations would still be capable of producing a vast number of offspring. Consequently, a population of adults could be reduced without affecting recruitment into that population during the following year. In the case of seabirds, however, a major reduction in the size of the breeding population also results in a proportional reduction in their reproduction and recruitment rates. It is therefore likely that marine fish populations could recover much more quickly from a severe reduction in abundance than could seabird populations.

The low recovery potential of arctic populations is a common theme in the publications of several authors discussing the biological implications of oil spills in the Arctic (e.g. Clark and Finley 1977, 1984; Sanborn 1977; Percy 1982). Although this may be true in certain groups of marine organisms (e.g. seabirds, marine mammals), it is unlikely to be the case with fish since maturation rates, longevity and fecundities of arctic species are not substantially different from those of temperate species.

CHAPTER 4

OIL IN BENTHIC ENVIRONMENTS

The following sections provide an overview of the behaviour and physico-chemical fate of oil reaching benthic sediments, and summarize existing information regarding the effects of petroleum hydrocarbons on benthic algae, infaunal and epifaunal invertebrates, and micro-organisms. The role of some micro-organisms in the degradation of hydrocarbons is also discussed, although these processes occur to a certain extent in each of the habitat types addressed in this report. However, most of the available information regarding the microbial degradation of oil has resulted from studies of benthic sediments.

Other biological resources not discussed in this chapter may also be affected by oil in benthic environments at certain times. These include demersal (bottom-dwelling) fishes and some species of marine mammals that are known to forage extensively (e.g. bearded seal) or occasionally (e.g. bowhead whale) on benthic prey. Some species of fishes also deposit spawn in or on benthic sediments. Those species of zooplankton that undergo extensive diel vertical migrations may approach the seafloor at certain times of the day. In fact, at water depths less than 50-80 m, all components of the planktonic community may be located near benthic environments during certain periods, although most zooplankton and almost all phytoplankton are distributed in the upper 25 m of the water column.

SECTION 4.1

OVERVIEW OF PHYSICAL AND CHEMICAL PROCESSES

Dave Green and Blair Humphrey

Oil may reach the sea floor through a number of different pathways. Some refined oils and greases are sufficiently dense to sink to the sea floor shortly after they are spilled on the water surface, and arrive at the bottom in a 'fresh' or unweathered form. These oils are composed primarily of high molecular weight hydrocarbons and will not change significantly in chemical composition with time because they are resistant to weathering processes such as microbial degradation and dissolution. Other types of oil, such as some crudes and bunker fuels, increase in specific gravity (and viscosity) as they weather and low molecular weight hydrocarbons are lost through evaporation and dissolution processes (Fingas et al. 1979), and may also eventually sink to the sea floor. Still other oil on the water surface and dispersed in the water column will reach the bottom through adsorption to suspended particulate matter (particularly clays) that eventually sinks. The rate of sinking depends on a number of factors including the sediment load of the water, the nature of the suspended sediments and the amount of oil adsorbed to particulates in the water column. Oil reaching the sea floor in this manner may be weathered or fresh. In addition, dissolved or dispersed oil may directly contact subtidal sediments, where it is then sorbed on the surface of sediment particles or entrained into interstitial water within the sediments. Oil dispersed in the water column may also reach the bottom within the fecal pellets of zooplankton. Some species of herbivorous zooplankton do not discriminate emulsified oil from other particulate matter. Oil ingested by these organisms, passes through the gut and is encapsulated within fecal pellets (Conover 1971). Oil stranded on shorelines may also be mixed with beach materials, increasing its specific gravity to the point where it will sink in adjacent subtidal habitats when transported off the beach by tidal and wave action.

Hydrocarbons in the bottom sediments may be subjected to different biogeochemical regimes depending on the absorption characteristics of the sediments (Meyer 1972). Fine-grained clays adsorb the largest quantities of hydrocarbons; this is due at least partly to the 3-5 percent organic content of clays. The availability of nutrients also affects the fate of oil reaching benthic environments, and tends to vary with sediment type. For example, sand and pebble sediments do not retain organic nutrients to any significant degree, and therefore weathering of oil can be very slow in areas where these types of sediments predominate. Oxygen levels also affect the rate of microbial degradation of oil and decrease with increasing depth into the sediments. Oil reaching the sea floor may be re-distributed due to natural reworking of the sediments caused primarily by bottom currents. These processes can also result in burial of oil present at the sediment-seawater interface, and exposure of buried oil in other areas.

Most weathering of oil occurs in the upper layer of the sediments; buried oil weathers extremely slowly. The most significant weathering processes in benthic environments are dissolution of water-soluble hydrocarbons and some by-products of microbial degradation into the water column, slow aerobic biodegradation by oleoclastic bacteria and, in some cases, very slow anaerobic microbial degradation. When combined, these processes result in the formation of naphthenic and naphtheno-aromatics, pentacyclic triterpanes, alkylated phenanthrenes and dibenzothiophenes, and an unresolved complex mixture (Boehm et al. 1982).

SECTION 4.2

BENTHIC ALGAE

Linda Martin

Several reviews of the effects of oil on algae and macro-algae have been published (Baker 1970; Johnson 1977; Snow 1982; Thomson and McLaren 1982), although none of them specifically addressed the effects of oil on benthic algae. Duval et al. (1981) summarized studies of the effects of past oil spills on marine plants. Despite these sources of information, the available information regarding the effects of oil on benthic macro-algae is very limited, and there are no data describing the effects of oil on benthic micro-algae in cold marine environments. Some of the information regarding the effects of oil on intertidal algae (Section 6.2) and planktonic algae (Section 3.2) may be relevant to benthic species, and is therefore cross-referenced throughout the following sections.

4.2.1 Mortality

Mortality of benthic macro-algae was reported following only seven of the 100 oil spill case histories reviewed by Duval et al. (1981). However, this may be partly a reflection of the interests of the investigators involved in these followup studies, rather than an indication of the frequency of mortality of benthic macro-algae due to oil spills. It is also possible that mortality of benthic algae may have occurred without being detected following some spills.

Encrusting coats of oil on kelp fronds were reported as the cause of mortality following a bunker fuel spill off Cape Flattery, Washington (Clark et al. 1975). However, researchers reported that extensive mortality of kelp did not occur following the Santa Barbara blowout and the grounding of the TAMPICO MARU, despite the fact that kelp beds were adjacent to the sites of both spills (North et al. 1965; Foster et al. 1971). Studies conducted during the BIOS program indicated that neither oil in sediments nor in the water column affected the biomass of the dominant benthic algal species (Cross et al. 1984).

There are no available data describing the acute lethal concentrations of oil to benthic macro-algae, although the effects of various oil concentrations and exposure durations on growth, photosynthesis and biomass of algal species have been examined (the results of these studies are summarized in later sections). North et al. (1965) reported that 3-h exposure to boiler fuel at a concentration of 1000 ppm was toxic to apical meristematic and vegetative blades of Macrocystis. Pneumatocysts (floats) of dead blades were filled with water, indicating that the oil had destroyed the integrity of the pneumatocyst walls. However, the toxic mechanism(s) responsible for mortality of these algae are largely unknown.

4.2.2 Contamination and Physical Damage

There are few detailed reports of contamination or physical damage of benthic macro-algae due to oil spills. Unspecified damage to kelp fronds has been reported following several spills (Duval et al. 1981). Accumulations of dead kelp have been observed along high tide lines following spills, although investigators were unable to determine if oiling occurred on the shore or at sea.

Kelp may be dislodged by the added weight of oil adhering to the fronds. However, Foster et al. (1971) found little damage to kelp beds which received significant amounts of oil during the Santa Barbara blowout. These researchers suggested that oil was prevented from adhering to the fronds by the mucilaginous excretions of the algae, and reported that oil was found only on damaged areas of plants and not on healthy fronds.

The only experimental data regarding the physical effects of oil on benthic algae are those from studies conducted by North et al. (1965) on the effects of diesel and boiler fuels on *Macrocystis*. Fronds of this alga became flaccid and lost turgor after 3-day exposures to concentrations of diesel fuel greater than 10,000 ppm. However, fronds exposed to 1000 ppm of diesel fuel for 3 to 6 h and then cleaned and immersed in new seawater did not exhibit loss of turgor. The investigators found that loss of turgor in *Macrocystis* blades was an early symptom of injury, and suggested that this effect confirmed the results of earlier studies (Overbeck and Blondeau 1954, cited in North et al. 1965) which suggested that oil acted first on the fatty constituents of the cytoplasmic membranes.

4.2.3 Photosynthesis and Production

All investigations of the effects of oil on photosynthesis in benthic macro-algae have shown that photosynthesis is reduced during oil exposure, although the degree of inhibition varies with oil type, concentration, length of exposure and the species of alga and the type of cells tested. Table 4.2-1 summarizes the available data on the effects of various types and concentrations of oil on photosynthesis in benthic macro-algae. Hsiao et al. (1978) reported a maximum inhibition of photosynthesis of 20 percent following a 4-8-h exposure of *Phyllophona* to 4 ppm of crude oil, while research by North et al. (1965) indicated that maximum inhibition of photosynthesis may occur after periods of brief oil exposure. The latter authors reported that *Macrocystis* exposed to 10,000 ppm of diesel fuel for periods of 3-6 h showed increasing inhibition of photosynthesis during the following seven days when transferred to clean sea water. In addition, these researchers noted the only measured increase in photosynthesis of a macro-algal species due to oil exposure. Photosynthesis in the bottom blades of kelp was stimulated by 24-h exposure to diesel fuel, but was inhibited by longer exposures.

TABLE 4.2-1

REDUCTIONS IN PHOTOSYNTHETIC CAPABILITY OF BENTHIC MACRO-ALGAE DUE TO EXPOSURE TO OIL

Oil Concentration (ppm)	Oil Type	Duration of Exposure (h)	Benthic Algae	Percent Inhibition of Photosynthesis	Ref*
3	Norman Wells	4 to 8	*Phyllophora*	2	1
4	Atkinson Point	4 to 8	*Phyllophora*	15	
4	Venezuelan	4 to 8		20	
43	Norman Wells	4 to 8	*Phyllophora*	25	1
43	Norman Wells	4 to 8	*Laminaria*	25	
83	Venezuelan	4 to 8	*Phyllophora*	20	1
83	Venezuelan		*Laminaria*	45	
100	Light Nigerian	24	*Acrosiphonia sonderi*	48	2
100	Light Nigerian	24	*A. arcta*	40	
100	Light Nigerian	24	*A. centralis*	16	
147	Atkinson Point	4 to 8	*Phyllophora*	25	1
147	Atkinson Point	4 to 8	*Laminaria*	25	
1,000	Diesel	72	*Macrocystis* (bottom blade)	62	3
1,000	Diesel	72	*Macrocystis* (surface blade)	22.5	
4,000	Crude Oils(4)**	4 to 8	Laminaria	45-62	1
4,000	Crude Oils(4)**	4 to 8	*Phyllophora*	25-45	
10,000	Diesel	3	*Macrocystis*	24.5	3
10,000	Diesel	6	*Macrocystis*	32	
10,000	Diesel	72	*Macrocystis*	100	

* References:
1 Hsiao et al. 1978
2 Kusk 1980
3 North et al. 1965

** Pembina, Atkinson Point, Norman Wells and Venezuelan

Thin films of oil applied directly to algal fronds have also inhibited photosynthesis in Macrocystis and Laminaria (North et al. 1965; Schramm 1972, cited in Johnson 1977). Johnson (1977) concluded that oil films reduced CO_2 exchange in macro-algae by lowering gaseous diffusion rates and exerting a toxic effect on photosynthesis. In summary, the above studies indicate that inhibition of photosynthesis occurs in some species of benthic macro-algae following brief exposures to low concentrations of oil, and that further reductions in photosynthesis may occur beyond the period of oil exposure.

4.2.4 Growth and Reproduction

The only information regarding the effects of oil on the growth and reproduction of benthic macro-algae is from the BIOS project. The effect of a surface oil slick biomass of three species of the brown algae were examined, but no significant changes were attributed to oil in the water or in the sediments (Cross et al. 1984). Some of the effects of oil observed in intertidal species (Section 6.2) may also occur when benthic algae are exposed to petroleum hydrocarbons. These include: (1) proliferation of tumour-like growths (Boney 1974); (2) morphological changes and premature fruiting (Boney 1974); (3) impeded development of reproductive products (Johnston 1977); and (4) inhibited growth of sporelings (Thomas 1977). North et al. (1965) also found that meristematic blades of Macrocystis were more sensitive than vegetative blades to diesel fuel exposure, suggesting that sensitivity may vary within individual plants.

4.2.5 Uptake and Depuration

The fate of oil contacting benthic algae is unknown. If oil is present on the surface of macro-algae, it would probably biodegrade rapidly due to the availability of oxygen in sea water and nutrient-containing secretions exuded from many species of algae. The potential for uptake of petroleum hydrocarbons by benthic macro-algae has not been examined.

4.2.6 Physiology and Metabolism

Studies of intertidal marine algal species (Section 6.2) suggest that a variety of effects may result from oil exposure including: reduced RNA and DNA synthesis (Davavin et al. 1975); accumulation of metabolic end-products (Baker 1970); disruption of chloroplast membranes (Baker 1970); and reduced CO_2 exchange (Schramm 1972, cited in Johnson 1977).

4.2.7 Habitat Loss and Disturbance

Oil may accumulate in benthic sediments following oil spills through the pathways described earlier in Section 4.1. Benthic macro-algae could be affected by oil on the seafloor if sporophytes did not settle or anchor on oiled rocks. However, this potential effect has not been documented. Oil in the upper layers of the sediments would be available to micro-algae, although there are presently no data describing the effects of oil on cold water marine micro-algae. Some studies with fresh water species suggest that a net reduction in photosynthesis may be followed by blooms of oil-tolerant species such as blue-green algae (Bott et al. 1976).

4.2.8 Indirect Effects

The only reported indirect effect of oil spills on benthic macro-algae have been an increased abundance of the local population of some species due to mortality of invertebrate grazers (Duval et al. 1981). This effect has been more frequently reported in intertidal algal communities, although the spatial extent of Macrocystis beds increased during the ten years following the TAMPICO MARU diesel spill, likely due to loss of sea urchins (North et al. 1965).

4.2.9 Effects of Countermeasures

During the BIOS Program studies, chemically dispersed oil had no effect on the biomass of dominant algal species (Cross et al. 1983). Mortality of benthic algae has occurred following oil releases whether dispersants were used or not (Duval *et al*. 1981). However, most investigators have attributed mortality of benthic algae to chemical (toxicity) or physical (dislodging) effects of oil. The potential contribution of dispersants to the impacts of past spills on macro-algal species has not been well documented (Duval et al. 1981).

4.2.10 Vulnerability

Species of benthic macro-algae with fronds that float on the water surface (e.g. *Macrocystis* and *Nereocystis*) may be vulnerable to oil slicks, as well as to dissolved or emulsified oil within the water column. Since reduced photosynthetic capacity of benthic algae has been reported following frond contamination, these species may be most vulnerable to oil spills. While laboratory studies have indicated that brief exposures (3-6 h) to relatively low concentrations (3-4 ppm) of dispersed oil can reduce photosynthesis of benthic algae, adverse effects on these communities have not been documented following experimental oil releases or past accidential spills. However, as indicated earlier, the lack of documented effects may reflect the research interest of investigators and the difficulties associated with the methods for detection of both short- and long-term oil spill impacts in subtidal environments. Oil may accumulate in benthic environments through a number of pathways, and benthic algae would be very vulnerable to effects of oil in these habitats.

SECTION 4.3

BENTHIC INFAUNA

Linda Martin

This section summarizes the biological fate and effects literature for infaunal invertebrates, a relatively immobile group of organisms that occur primarily within the bottom sediments. Bivalve molluscs and polychaetes generally account for much of the infauna biomass, and these organisms have also been the focus of most studies of the effects of hydrocarbons on benthic infauna. Less numerous infauna taxa, such as Holothuroidea and Ophiuroidea, are infrequently discussed in the oil effects literature. Major information sources for this section include; a recent review of the effects of oil on arctic marine invertebrates by Wells and Percy (1985), investigations by the Auke Bay Laboratory between 1970 and 1981 summarized by Rice et al. 1983, earlier reviews of hydrocarbon effects on arctic and subarctic marine organisms by Craddock (1977); Johnson (1977); Thomson and McLaren (1982); Percy (1982), a review of oil spill case histories (Duval et al. 1981) and results of the BIOS project. The comprehensive BIOS studies which included spatial and temporal controls for experimental field releases of oil and chemically treated oil, as well as supplimentary laboratory investigations, provides substantial new information on hydrocarbon effects on shallow water benthic invertebrate communities and individual organisms.

4.3 Mortality

The potential for deposition and sinking of oil to the sea floor and the persistence of oil in bottom sediments (Section 4.1), combined with the relative immobility of infauna, suggest that this group is particularly vulnerable to the effects of hydrocarbons. When high concentrations of oil have accumulated in benthic sediments as a result of oil spills, mortality of benthic infauna has generally been greatest (Duval et al. 1981), and investigators have frequently found positive correlations between the degree of mortality of benthic organisms with oil concentrations measured in sediments. For example, Sanders (1977) documented near total mortality of benthic organisms at stations with highest fuel oil concentrations (40-450 mg L^{-1}) after the FLORIDA spill, and relatively little mortality at stations with 2.5 to 8.0 mg L^{-1} hydrocarbons. Mortality of benthic infauna has been reported following numerous other oil spills, such as the AMOCO CADIZ, ZOE COLOCOTRONIS, TAMPICO MARU, USAF Searsport and Deception Bay incidents (Duval et al. 1981). However, lack of supporting data regarding oil fate, exposure durations, and sediment and water column hydrocarbon concentrations have made it difficult to reach firm conclusions regarding the mechanism and hydrocarbon concentrations responsible for these mortalities.

Laboratory investigations provide some information regarding mortality of infauna following exposure to various oil concentrations within sediments (Table 4.3-1). Data presented in this table indicate that significant mortality of infauna due to crude oil exposure occurs when oil-in-sediment concentrations exceed approximately 400 mg L^{-1}, although some species are tolerant of this and higher concentrations. Species tolerance may be linked to the mode of feeding of different groups of organisms. Augenfeld et al. (1980) found that the deposit feeder Macoma inquinata accumulated more oil and suffered greater mortality than the filter feeder Protothaca staminea. Other factors, such as oil type, degree of weathering that occurred prior to deposition, and the fate of oil within bottom sediments, are important determinants of the effects of oil on infauna. Refined fuels, such as the No. 2 fuel released during the FLORIDA spill, resulted in significant mortality at concentrations as low as 40 mg L^{-1} (Sanders 1977), and fresh oil that is buried and unavailable for microbial degradation may result in long-term effects on infauna.

While filter feeders may be more resistant to oil-in-sediment, they are susceptible to oil in the water column. Table 4.3-2 summaries some of the available acute lethal toxicity data for infauna exposed to water

TABLE 4.3-1

ACUTE LETHAL TOXICITY OF OIL IN SEDIMENTS TO INFAUNA INVERTEBRATES
(Adapted from Thomson and McLaren 1982)

Species	Oil Type*	Concentration (mg L^{-1})	Lethal Effects	Reference
Macoma balthica	PB	0.5 cm contaminated sed. 1 part oil: 2 parts dry sed.	Significant mortality	Taylor et al. 1976
	PB	0.6 cm contaminated sed. 1 part oil: 2 parts dry sediment	50 percent burrow to surface	
Macoma inquinata	PB	1144 decreased to 364	41 percent mortality in 38 days	Roesijadi and Anderson 1979
Macoma inquinata	PB	1237 decreased to 850	83 percent mortality in 54 days	Augenfeld et al. 1980
Protothaca staminea	PB	1237 decreased to 850	83 percent mortality in 54 days	Augenfeld et al. 1980
Abarernicola pacifica	PB	251	10 percent mortality in 11 days	Augenfeld 1980
	PB	415	20 percent mortality in 11 days	
	PB	807	30 percent mortality in 11 days	

*Prudhoe Bay Crude Oil

TABLE 4.3-2

ACUTE LETHAL TOXICITY OF THE WATER SOLUBLE FRACTION OF PETROLEUM HYDROCARBONS TO INFAUNA INVERTEBRATES

(Adapted from Wells and Percy 1985 and Thomson and McLaren 1982)

Species	Oil Type*	Concentration (mg L^{-1})	Lethal Effects	Reference
Capitella capitata	2FO BC SL K	2.3 0.9 12.0 9.8	96-h TLm 96-h TLm 96-h TLm 96-h TLm	Rossi et al. 1976
Protothaca staminea	PB	>14.7	96-h TLm	Rice et al. 1976
Macoma balthica	PB	0.036	20 percent mortality in 11 days	Taylor et al. 1976
	PB	0.019-0.302	0 percent mortality in 11 days	
Neanthes arenaceodentata	SL	12.0	96-h LC_{50}	Rossi and Anderson 1976
Chalamys hericus	CI	3.94	96-h LC_{50}	Rice et al. 1979
Tapes semidecussata	CI	35.0	96-h LC_{50}	Nunes and Benville 1978
Macoma balthica	PB	0.3	8 percent mortality in 180 days	Stekoll et al. 1980
		3.0	81 percent mortality in 180 days	

* BC - Bunker C
CI - Cook Inlet Crude Oil
PB - Prudhoe Bay Crude Oil
2FO - No. 2 Fuel oil
SL - South Louisiana Crude Oil
K - Kuwait Crude Oil

soluble fraction of various oils in laboratory experiments. Although data summarized in Table 4.3-2 indicate that some infauna are sensitive to low concentrations of a WSF, other investigators have been unable to determine 96-h LC_{50} values for test organisms (Thomson and McLaren 1982). For example, Rice et al. (1979) could not ascertain the 96-h LC_{50} for 11 of 13 bivalves and two polychaetes tested during their research.

Experimental oil release during the BIOS program has not yet caused any significant change in terms of benthic community abundance or biomass (Cross et al. 1984). The fate of the oil has probably been an important determinant of the lack of effects. Initially, concentrations of oil under the slick were low and confined to the upper meter of the water column, and direct penetration of oil into subtidal sediments was minimal (Sergy, in press). However, oil stranded in the intertidal zone was eroded into subtidal sediments during the years following the release, and concentrations of oil in subtidal sediments two years after the spill ranged from several hundred mg L^{-1} in shallow subtidal areas to a few mg L^{-1} approximately 500 m offshore. This long-term exposure to relatively high concentrations of oil is an area of concern. Fresh crude oil at concentrations within this order of magnitude were sufficient to cause mortality of some polychaetes (Table 4.3-1), and the more toxic No. 2 fuel oil caused significant mortality of macrofauna in an experimental ecosystem following 25-week exposures to oil-in-sediment concentrations of 109 mg L^{-1} (Grassle et al. 1980).

4.3.2 Growth, Development and Reproduction

Reported adverse effects of the WSF of oil on various aspects of the growth, development and reproduction of benthic infauna have included reduced weight-length relationships, reduced condition indices, and abnormal cell development and differentiation in larval stages. Hutcheson and Harris (1982) found that the Scope for Growth (assimilation relative to respiration rates) in clams (Mya and Astarte) was reduced at a nominal fresh oil concentration of 0.8 mg L^{-1} and a weathered oil concentration of 0.95 mg L^{-1} (WSF, 4-day exposure), respectively. Renzoni (1975, summarized in Johnson 1977) found that fertilization and normal development of both *Mulinia lateralis* and *Crassostrea angulata* and growth of *M. lateralis* larvae were adversely affected by crude oil at concentrations ranging from 1 to 1000 ppm (WSF).

Wells and Percy (1985) note that there are few studies of the effects of oil on the growth of arctic benthic organisms, probably due to slow growth rates of polar species and limited sampling season. Stekoll et al. (1980) reported that *Macoma balthica* exposed to 0.03-3.0 mg L^{-1} crude oil for 180 days had reduced growth rates and a negative energy budget. These investigators also found evidence of abnormal gonad development in the bivalves exposed to the lowest oil concentration (0.3 mg L^{-1}) for four months.

The effects of oil-contaminated sediments on polychaetes appears to be relatively species-specific. Percy (1982) cites several investigations that document reduced growth of some juvenile and larval polychaetes due to oil exposure. Other polychaete species appear to be resistant to petroleum hydrocarbons in sediments (Busdosh et al. 1978). *Capitella capitata* reproduced very successfully following the FLORIDA spill, and became dominant in areas with extensively oiled sediments where other species were killed (Sanders et al. 1980). Similarly, Busdosh et al. (1978) reported that *Nephytes* sp. preferred oiled to clean sediments. However, other investigators have demonstrated significant reductions in the fecundity of polychaetes that had been exposed to hydrocarbons (Akesson 1975 and Carr and Keish 1977, cited in Percy 1982). Cross et al. (1984) also found that in oil-contaminated sediments, reproduction of one year class of the polychaete *Spio* sp. was inhibited following the BIOS experimental oil spill.

Growth of bivalves has been adversely affected by oil-contaminated sediments in field exposures. Follow-up studies of the ARROW spill indicated that growth of *Mya* was still reduced (weight gain per unit length) six years after the event (Thomas 1978), and Dow (1975) also found evidence that growth and production of Mya was reduced in oiled sediments. Oil released during the BIOS study affected length-weight relationships of Macoma calcarea (Cross et al. 1984). This bivalve is a deposit feeder and high concentrations of oil were accumulated in its body tissues from oil-contaminated sediments during the two weeks following the experimental spill.

4.3.3 Metabolism and Physiology

In recent years, increasing emphasis has been placed on the examination of metabolic and physiological functions in oil spill effects research, but the available data are still relatively limited. Gilfillan (1975) and Gilfillan et al. (1976) attributed reductions in carbon flux in Mya following oil exposure to reduced feeding rates and assimilation efficiencies, coupled with higher respiration rates. Gilfillan and Vandermeulen (1978) later suggested that altered respiratory demands were probably the lesser of the two factors contributing to reduced carbon flux. Stainken (1976) observed increased respiration rates in Mya after exposure to $\leq$ 50 mg L^{-1} of No. 2 fuel. Similar changes in oxygen consumption have also been documented in benthic epifauna (Section 4.4.3). Percy and Mullin (1975) investigated feeding rates in the arctic bivalve Yoldiella intermedia and observed an initial cessation of feeding (followed by a resumption). They also noted that the duration of the non-feeding period was directly proportional to the oil concentration. However, Anderson (1972, cited in Johnson 1977) found that filtration rates of Mya increased during exposure to 0.9 percent Bunker C fuel. Percy (1982) noted inconsistencies in the results of past studies on the effects of different oil concentrations with the same species in various experimental situations. Nevertheless, he emphasized that sufficiently large or prolonged alterations in metabolic rate could be considered critical in terms of the physiology of affected individuals.

Dillon (1984) summarized studies of the effects of oil on the enzyme systems and biochemical composition of marine organisms. He cited the research of Clement et al. (1980) and Stekoll et al. (1980) which indicated that 6-month exposure of marine bivalves to 0-3000 ppm crude oil had no effect on Mg^{++}, Na^{+} or K^{+}ATPase activities. However, other studies summarized by Dillon (1984) indicated that there were reductions in total free amino acids (alanine, lysine, taurine, aspartic and glutamic acids) following exposure of bivalves to oil-contaminated sediments (Augenfeld et al. 1980; Roesijadi and Anderson 1979).

Wells and Percy (1985) cite long-term studies, involving chronic oil exposure that have resulted in a stimulation of metabolism in bivalves leading to negative energy balance, and reduced shell growth, tissue weight, nutrient storage and gonad development (Dunning and Major 1974). Wells and Percy (1985) suggest that because of the range of physiologically active compounds in crude oil, complex metabolic responses to oil exposure are to be expected.

4.3.4 Uptake and Biotransformation

The rate of uptake and depuration of oil ingested by benthic infauna and epifauna depends, in part, on their feeding habits and processes affecting the fate of the oil in the environment. Since oil does not persist in the water column for long periods, it is taken up and depurated relatively quickly by filter feeders, including many bivalves and some tube-dwelling polychaetes, usually with little degradation (Boehm et al. 1982, 1984). On the other hand, deposit/detrital feeders such as echinoderms, many polychaetes, and some bivalves, ingest oil present within the sea floor (Roesijadi et al. 1978). These fauna may be chronically exposed to oil in benthic habitats, and any oil ingested during feeding may be metabolized.

The rate of hydrocarbon uptake by infauna and the mobilization of hydrocarbons within organisms was examined in laboratory and field investigations during the BIOS project, as well as by a few earlier laboratory studies. Gilfillan and Vallas (1984) found that Serripes groenlandicus (a fast filter feeder) accumulated hydrocarbons more quickly and to a greater degree than the slower filtering Mya truncata. There is some evidence for differential uptake of hydrocarbon compounds among tissues within the same organism. For example, Boehm (1983) found that the more water-soluble aromatics (napthalene, alkylated benzenes) are transported to the muscle tissues rapidly in S. groenlandicus, while the less water-soluble aromatics (phenanthrenes and bibenzothiphenes) were preferentially located in the gut. In addition, there may be selective uptake of certain hydrocarbons from the gut of some molluscs. Vandermeulen et al. (1981) reported limited uptake of aromatics with five or more rings in some molluscs following the AMOCO CADIZ spill. Fucik and Neff (1977) reported temperature-dependent uptake of naphthalenes in two clams. Uptake was higher in colder waters, but depuration was not affected by temperature.

Ingested hydrocarbons are usually depurated rapidly in the faeces. The half-life of oil in the gut of invertebrates exposed to oil for short periods ranges from a few days to a few weeks. During BIOS field studies, filter feeders were found to depurate 60-75 percent of hydrocarbon body burdens within approximately two weeks (Boehm 1983). Stainken (1977) also found that 50 percent of tissue hydrocarbons were depurated after 14 days. Alkanes were depurated more quickly than aromatics and low molecular weight aromatics were depurated faster than polynuclear aromatics. Anderson (1975) showed that unless the benthic fauna are persistently exposed to oil in the sediments or the overlying water column, depuration of accumulated hydrocarbons is rapid. Where benthic invertebrates are chronically exposed to oil, they may accumulate hydrocarbons (by feeding in contaminated sediments or by direct absorption through the gills and body walls) and eventually store them and their metabolites in lipid deposits (Wolfe et al. 1981). There is little evidence that biomagnification occurs when fauna containing hydrocarbons are ingested by predator species (Varnasi and Malins 1977).

4.3.5 Pathology and Histology

Abnormal tissue growth and lesions have been suggested as possible effects of oil exposure (Percy 1982). However, there is no information in the literature on the pathological and histological effects of petroleum hydrocarbons on cold-water species of benthic infauna.

4.3.6 Behaviour

Behavioural responses of polychaetes and bivalves to moderate to high oil contamination in benthic sediments include: (1) leaving their burrows; (2) slow response to physical external stimuli; and (3) inhibition of sediment working and cast production (Dow 1975; Gilfillan 1975; Prouse and Gordon 1976; Gilfillan et al. 1976). Prouse and Gordon (1976) also found no effects of oil exposure on the behaviour of polychaetes at concentrations of ≤ 25 µg g^{-1} sediment. Diver observations of benthic infauna beneath a surface oil slick during the BIOS project indicated no immediate behavioural responses of infauna to the oil (Cross and Thomson 1982). Although, oil from the surface slick did not disperse throughout the water column or directly reach sublittoral sediments, oil stranded on the shoreline was eroded and accumulated in shallow nearshore areas in subsequent years. Effects of hydrocarbons on the behaviour of infauna have yet to be documented in oil-contaminated (several hundred mg L^{-1}) sediments at the BIOS site.

Laboratory studies also indicate that adverse behavioural effects can result from exposure to oil WSFs. Taylor et al. (1976, cited in Johnson 1977) reported that Macoma balthica moved to the sediment surface within three days during exposure to 0.33 ppm oil, and within nine days at 0.036 ppm (WSF expressed as naphthalane equivalents, static bioassay). During flowthrough bioassays described by Stirling (1975), phenol concentrations of 10-500 ppm caused the surfacing of some clams and inhibited the burrowing response of others. Some clams resumed normal activities when they were returned to clean seawater.

4.3.7 Habitat Loss and Disturbance

The foregoing indicates that some benthic infauna are susceptible to high concentrations of oil in the sediments, but may tolerate low oil concentrations in sediments. Other benthic genera are capable of tolerating relatively high oil concentrations (e.g. Capitella) and populations of these organisms have increased in the absence of competitors. However, in general, significant sublethal effects and mortality of infauna can be expected if unweathered crude oil reaches and persists in bottom sediments in concentrations exceeding approximately 250 mg L^{-1}. Even lower concentrations of the more toxic refined fuels and some crude oils in bottom sediments would also be of concern. In addition, infauna are sensitive to relatively low concentrations of oil dispersed in the water column (Table 4.3-2). If natural processes or use of chemical dispersants result in mixing of oil throughout the water column, adverse effects on infauna are also likely to result. As indicated in Section 4.1, this habitat is susceptible to oil exposure through a number of pathways, and some of these pathways may produce delayed effects on benthic communities following oil spills (e.g. erosion of oil stranded in intertidal areas slowly accumulating in subtidal sediments).

4.3.8 Indirect Effects

The most probable indirect effect of oil on the infaunal benthic community would be predation when polychaetes or bivalves migrate to the sediment-water interface or leave their burrows. The significance of this effect would undoubtedly depend on several factors, including the abundance of predators and the effects of oil on these organisms, the duration of exposure of infauna to predators and their ability to rebury themselves, and oil concentrations in benthic habitats and overlying waters.

A change in composition of the community could result if mortality of large numbers of infauna occurred and oil-tolerant genera (e.g. Capitella) colonized the area. This indirect effect may be only a short-term phase in the recovery process, since in other cases, such opportunistic species have peaked in numbers and then decreased as community structure returned to normal (Sanders et al. 1980).

4.3.9 Effects of Countermeasures

Oil spill countermeasures that increase oil sedimentation rates or cause oil to disperse throughout the water column may increase oil-infauna contact and therefore oil-related effects. Since there appears to be no information regarding sinking agents or emulsifiers and their effects on infauna, this section deals exclusively with the effects of chemically dispersed oil on infauna. The more recently developed dispersants are relatively non-toxic substances, although chemically dispersed oil itself is acutely toxic to infauna. Mageau and Engelhandt (1984) found that a chemically dispersed (Corexit) oil concentration of 500 mg L^{-1} was lethal to *Serripes groenlandicus* in a 6-day exposure, and Foy et al. (1980) reported a 96-h LC_{50} for *Mya arenaria* with dispersed crude oil of 173 to 205 mg L^{-1}.

The effects of chemical dispersion of oil on arctic infauna were assessed in both field and laboratory investigations during the BIOS project and described in Cross and Thomson (1982); Mageau and Engelhardt (1984); Gilfillan and Vallas (1984); and Cross et al. (1984). Field observations indicated a greater immediate effect of dispersed oil *vs* non-dispersed oil on the behaviour of benthic infauna (Cross and Thomson 1982). This was because dispersed oil penetrated bottom sediments causing initial oil-in-sediment concentrations of 1 to 5 mg L^{-1} in the upper few centimeters, while at the same time, oil was present in the water column at concentrations of 1 to greater than 50 mg L^{-1} (Boehm 1983). In the areas affected by chemically dispersed oil, large numbers of bivalves and polychaetes moved to the sediment surface and exhibited abnormal behaviour (e.g. gaping, lack of or delayed response to physical stimulation, ostial closure and siphon retraction). Conservative estimates of the relative biomass of *Serripes* on the sediment surface ranged from 28 to 36 percent of pre-spill subsurface densities. Within a few days following the spill, densities of *Serripes* on the sediment surface decreased to 9 percent and investigators suggested that predation and/or reburial may have occurred. Minimal long-term effects of oil on the infaunal community have been detected during two years of follow-up study, and have not been reflected in significant changes in community structure or abundance. Field investigations conducted to date have documented some minor and temporary effects, including: (1) size-specific mortality of *Macoma* juveniles; (2) increase in the abundance of *Astarte* juveniles; (3) decrease in tissue weight per length in *Serripes*; and (4) inhibition of reproduction of the 1981 year class of *Spio* (Cross et al. 1983; Cross et al. 1984). It should be emphasized, however, that these effects did not result in detectable changes at the population or community level during two years of post-spill monitoring.

The BIOS project included a series of laboratory experiments that were designed to elucidate the behavioural responses of *Mya truncata* and *Serripes groenlandicus* observed in the field, and to correlate these responses with dispersed oil concentrations and exposures typical of actual oil spills. *M. truncata* showed fewer behavioural abnormalities in response to all dispersed oil concentrations and exposure durations than *S. groenlandicus*, and no clear dose-response relationships for behavioural indices could be established for this test organism. Chronic exposure (14 days) of *M. truncata* to 50 ppb and 500 ppb of dispersed oil resulted in no adverse behavioural responses. On the other hand, *S. groenlandicus* exposed to the same oil levels for similar durations initially responded by closing ostia, and then emerged from the sediment in a narcotized state. At dispersed oil concentrations of 50 ppm and higher, behavioural effects included siphon retraction and forceful shell closure. Ostial closure at all oil concentrations and mantle/shell gape observed at higher concentrations in *S. groenlandicus* would be important factors determining the degree of exposure of soft tissues to hydrocarbons (Mageau and Engelhardt 1984).

BIOS laboratory projects also investigated: (1) accumulation of hydrocarbons in tissues; (2) physiological effects of accumulated hydrocarbons; and (3) the effects of differences in filtration rate on hydrocarbon accumulation by Mya truncata and Serripes groenlandicus at various dispersed oil concentrations (50 and 500 ppb) and exposure durations (Gilfillan and Vallas 1984; Mageau and Engelhardt 1984). The major results of these investigations are summarized below. Low dispersed oil concentrations did not result in detectable dose-response relationships for Scope for Growth or enzyme activity in these invertebrates. However, low concentrations of dispersed oil did result in accumulation of trace amounts of hydrocarbons in tissues of the bivalves (Mageau and Engelhardt 1984). Depuration occurred in both species, and although it appeared to be independent of filtration rate, S. groenlandicus showed a greater relative retention of hydrocarbons in tissues than M. truncata. Exposure of the bivalves to changing oil concentrations that simulated field conditions resulted in clear dose-response relationships for body burdens of aromatic hydrocarbons and for Scope for Growth (assimilation, filtration and respiration rates). Accumulation of hydrocarbons in the tissues of M. truncata and S. groenlandicus was clearly related to oil concentrations and filtration rates of each species and, therefore, S. groenlandicus accumulated higher final body burdens of petroleum hydrocarbons than M. truncata (approximately 10:1). Activity of glucose-6-phosphate dehydrogenase and asparate amino-transferase, important enzymes in the metabolism of carbohydrates and proteins, respectively, was also affected by hydrocarbon exposure (Gilfillan and Vallas 1984).

Examination of uptake and depuration of dispersed oil by infauna from sediments and the water column during the BIOS program also included sampling of animals from sites of oil exposure in the field. The most significant findings of these studies have been summarized by Boehm (1983). When initial oil concentrations in filter feeding bivalves (M. truncata and S. groenlandicus) in sites contaminated by high dispersed oil concentrations were compared to animals from sites where no dispersants were used and oil concentrations in the water column were low, levels of hydrocarbons in tissues were equivalent or greater. These results contradict the laboratory findings discussed above, and Boehm (1983) suggested three possible explanations for similar tissue hydrocarbon concentrations despite exposure to different (order-of-magnitude) oil-in-water concentrations: (1) cessation of pumping or mortality at high oil concentrations; (2) inability to transport oil across membranes at rates sufficient to accumulate more oil; and (3) saturation of hydrocarbons in gut tissues. *M. truncata* and *S. groenlandicus* were also found to preferentially retain different hydrocarbon assembleges. *S. groenlandicus* retained high molecular weight saturated compounds, while *M. truncata* depurated all hydrocarbons. Tissue hydrocarbon concentrations in both these filter feeders decreased as oil quantities in the water column decreased. However, oil in tissues of deposit/detrital feeders (i.e. *Macoma* and some polychaetes) reflected the concentrations of oil in sediments, which increased with time at some sites.

Contradictory results have been obtained during other investigations that have compared the effects of chemically *vs*. physically dispersed oil on benthic infauna. Gilfillan et al. (1984, cited in Gilfillan and Vallas 1984) found no physiological effects of dispersed oil in *Mya arenaria* after a 7-day exposure, while Hutcheson and Harris (1982) reported that *Astarte borealis* was either more or less sensitive to dispersed oil than to water-soluble oil, depending on the concentration. Scope for Growth of this species decreased at concentrations from 1.6 to 1.8 mg L^{-1}, but at lower total hydrocarbon concentrations (<1 mg L^{-1}), this measure of net assimilation actually increased (Hutcheson and Harris 1982).

4.3.10 Vulnerability

The probability of contamination of shallow benthic habitats following of an oil spill is a relatively high, although a number of factors (Section 4.1) determine the concentration and nature of oil in benthic sediments. Deeper benthic environments may be less vulnerable to initial oil spill effects, but may eventually be contaminated to some degree by oil that is transported from shallow areas. Infauna are extremely susceptible to oil due to their sessile habits. Exposure can occur both through physical contact and ingestion of oil in sediments, as well as through contact with emulsified or dissolved oil in the overlying water column. Results of the BIOS study indicate that surface oil slicks that do not contaminate bottom sediments may have little or no significant long-term effects on benthic infauna. Brief exposure to moderate concentrations of chemically dispersed oil in the water column and sediments may result in short-term effects on organisms, but may not cause long-term changes in community structure.

4.3.11 Recovery

The time for recovery of benthic infaunal communities following exposure to petroleum hydrocarbons will depend upon a number of species-specific factors, as well as physical/biological processes responsible for the fate of oil in the environment. In general, recovery would require the longest period in areas where oil accumulates in benthic environments and the benthic community is comprised of detrital/deposit feeders. Filter feeders and some oil-tolerant species may become dominant members of the benthic community in these situations, at least until oil concentrations were reduced by either microbial degradation or physical processes. On the other hand, relatively rapid recovery may occur if oil is not deposited in benthic sediments or mixed into the water column to depths inhabited by filter feeders. Individual species characteristics, such as filtration rates and tendencies for retention of individual petroleum compounds may also assume a role in the rate of recovery. Insufficient information is available regarding these latter factors to determine their relative importance to the recovery of infaunal communities. There is some evidence that the application of chemical dispersants will not significantly affect recovery of benthic infauna following oil spills. Results of the BIOS studies conducted to date suggest that increased risks of temporary effects and availability of oil to subtidal fauna due to nearshore chemical dispersion, may be preferable in situations where shoreline protection is important (Sergy, in press).

SECTION 4.4
BENTHIC EPIFAUNA

Linda Martin

Epifauna include both sessile and motile members of the benthic community, but unlike infauna, these organisms are found on rather than within the sediments. Crustaceans and echinoderms are the predominant taxonomic groups that comprise the epifauna. The major information sources reviewed during preparations of this section are the same as those identified in the previous section on benthic infauna.

4.4.1 Mortality

Mortality of benthic crustaceans, including crabs, shrimp, lobsters and amphipods has been more frequently documented following oil spills than mortality of echinoderms (Duval et al. 1981). However, large numbers of sea urchins and/or starfish were reported to have washed ashore following some spills, such as the ZOE COLOCOTRONIS, AMOCO CADIZ and DONA MARIKA incidents (Duval et al. 1981). Mortality of epifauna has not been documented in all field research programs following actual or simulated oil spills. For example, the BIOS experimental oil spill did not result in reduced densities of *Strongylocentrotus droebachiensis*, *Leptasterias polaris* or 11 taxa of epibenthic crustaceans (Cross et al. 1984). Mortality appears to be highly dependent on the circumstances surrounding the events, and would probably be greatest when conditions favoured: (1) the sedimentation of oil in subtidal habitats; (2) the dissolution of hydrocarbons from contaminated intertidal sediments; and (3) the presence of high hydrocarbon concentrations in near bottom waters.

Few researchers have examined the relationship between the degree of oil contamination in benthic sediments and mortality of epibenthic invertebrates. Percy and Mullin (1975) investigated the toxicity of sediments contaminated with three different crude oils to Onisimus affinis under different temperature regimes. These authors reported that: (1) toxicity declined with time and mortality was less than 30 percent after 60 days; (2) different temperature regimes had no consistent influence on mortality; and (3) 50 percent of the test organisms died within 10 days when exposed to 0.25-1.0 mL of oil agitated with 10 g of sediment. Sublethal effects on epibenthic crustaceans exposed to the oiled sediments have been more frequently examined than acute toxic effects, and the results of these studies are discussed in subsequent sections.

The acute lethal toxicity of the WSF of oils to epibenthic invertebrates has been investigated in numerous laboratory research programs. The results of these studies have been summarized by Thomson and McLaren (1982); Rice et al. (1983); and Wells and Percy (1985). Such reviews have consistently emphasized the difficulties associated with comparisons among laboratory toxicity data due to differences in oil types, exposure regimes and methods of preparation of WSFs. Nevertheless, if the effects of different oil types are eliminated, some general trends for similar conditions have been recognized. Rice et al. (1983) concluded that Alaskan pelagic invertebrates were more sensitive to dissolved hydrocarbons than either benthic or intertidal Alaskan species (LC_{50} 1-3 ppm vs. >8 ppm, respectively). Wells and Percy (1985) concluded that despite the existence of large inter-specific variability in sensitivity, amphipods and decapods appeared to be more sensitive to oil than isopods. They also reported that acute lethal concentrations of oil with benthic organisms generally are less than 50 mg L^{-1} (measured hydrocarbon concentration). The sensitivity of some crustaceans (e.g. tanner crabs) to oil also appears to be increased during moulting (Karinen and Rice 1974; Rice et al. 1976) and when animals are already stressed (e.g. from increased salinity; Schneider 1980). The results of a number of studies evaluated by Rice et al. (1983) showed that the effects of temperature on the sensitivity of individual Alaskan species appears to be quite variable. These authors concluded that this variability was probably due to the combined effects of temperature on persistence of some petroleum hydrocarbons and on physiological responses.

In general, larval stages of invertebrates appear to be more sensitive than adults to oil exposure. Larval crustaceans are very sensitive to hydrocarbon exposure and concentrations as low as 0.2 ppm are lethal to coonstripe shrimp larvae (*Pandalus danae*, Rice et al. 1983), whereas lethal concentrations for adults are generally in the range from 1 to 10 ppm (Thomson and McLaren 1982; Wells and Percy 1985). Foy (1979) documented greater sensitivity of *Onisimus littoralis* juveniles compared to adults. Rice et al. (1983) also reported increased sensitivity of larval *vs*. adult stages of some Alaskan invertebrates, and stated that oil exposure may impede the moulting of larvae. On the other hand, exposure of gravid coonstripe shrimp to the WSF of crude oil for 28 days indicated that the eggs of this species were as tolerant as the adult females (Carls, in prep., cited in Rice et al. 1983). Juvenile isopods (*Mesidotea entomon*) were also found to be oil tolerant (0.2-0.9 mg L^{-1} WSF; Percy 1978).

4.4.2 Growth, Reproduction and Development

Sublethal effects of oil-in-water exposures on the reproductive processes of crustaceans that have been observed during laboratory studies include reduced rates of brood development (Wells and Sprague 1976), premature release of eggs (Busdosh et al. 1978), reduced frequency of precopulation (Linden 1976), reduced mating activity (Duval et al. 1982), and reduced brood size due to adverse effects of hydrocarbons on hatching rate (Tatem 1977). Oil concentrations (usually <1 ppm) and exposure periods (<72 h) necessary to produce these sublethal effects have been relatively low.

Fewer data are available on the sublethal effects of oil on reproduction of echinoderms. Adverse effects of hydrocarbons on cleavage and the development of sea urchin embryos have been documented (Allen 1971; Lonning and Hagstrom 1975, both cited in Johnson 1977, Falk-Petersen et al. 1983), but in some of these same studies, fertilization success has been shown to be relatively insensitive to even very high oil concentrations (20 000 ppm).

Falk-Peterson et al. (1983) found that oil weathered on the shore for 5 to 12 months was no longer toxic to sea urchin embryos during 4-day laboratory exposures at a maximum oil-in-water concentration of 1.6 ppm. However, this and lower concentrations of fresh oil resulted in greater than 80 percent mortality of embryos (Falk-Peterson et al. 1983).

Johnson (1977) concluded that no generalizations could be made regarding the effects of oil on the growth of marine crustaceans. Growth of larval amphipods and shrimp was reduced during exposure to low (< 1 ppm oil-in-water) concentrations of crude oil (Linden 1976; Tatem 1977). Percy (1978) found that low concentrations of weathered crude oil (0.52 ppm, 160-d exposure) slightly stimulated growth of isopods (*Mesidotea entomon*), while higher concentrations (1.01 ppm, 84- and 160-d exposure) inhibited growth. No consistent effects on growth of *M. entomon* due to exposure to several types and concentrations (1.71 to 0.04 ppm) of fresh crude oil were observed by Percy (1978). Duval et al. (1982) reported that growth of the isopod *Exosphaeroma oregonesis* was not significantly affected over an 8-wk recovery period following a short exposure (24 h) to 18 mg L^{-1} (WSF) of crude oil. The results of these studies support Johnson's (1977) view that growth may not be a good indicator of oil-related stress in isopods and perhaps other crustaceans.

Both the success and timing of moulting in crustaceans are affected by oil exposure. Wells and Sprague (1976) found that oil (> 0.13 mg L^{-1} physically dispersed) delayed moulting by early larval stages of lobsters. Tanner crabs (*Chionoecetes bairdi*) autotomized their limbs shortly after or during moulting (Karinen and Rice 1974). Duval and Fink (1980) and Duval et al. (1982) found that the frequency of moulting of the isopod *Exosphaeroma oregonesis* was increased by 12- to 48-h exposure to concentrations of petroleum hydrocarbons as low as 2.4 mg L^{-1} (WSF), but higher concentrations of oil (12.5 to 59.3 mg L^{-1}) reduced moulting. On the other hand, Percy (1978) found no evidence that oil exposure (1.7 mg L^{-1} WSF) inhibited moulting of the isopod *Mesidotea entomon*.

4.4.3 Physiology and Metabolism

Johnson's (1977) review of the sublethal effects of oil indicates substantial variability in the respiration rates of crustaceans in response to different oil types and concentrations during laboratory exposures. In only one study was oxygen consumption in a sea urchin (*Strongylocentrotus droebachiensis*) examined during oil exposure (Yentsch et al. 1973, cited in Johnson 1977), and the results of this investigation indicated that respiration rates were slightly increased. Osmotic regulation in some shrimps has been disrupted by oil exposure. Anderson et al. (1974, cited in Johnson 1977) found that the body burden of hydrocarbons was related to the extent of disruption, and that osmotic regulation returned to normal as depuration of hydrocarbons proceeded. As a result of eleven years of research at the Auke Bay Laboratories in Alaska, Rice et al. (1983) have concluded that long-term exposures (30-40 d) to sublethal concentrations of petroleum hydrocarbons (20-30 percent of the long-term LC_{50}) can affect the physiology of most species, reduce growth and reproduction and change behaviour. In addition, these authors suggest that longer exposure to even lower hydrocarbon concentrations may still have deleterious effects.

4.4.4 Uptake and Biotransformation

The uptake and biotransformation of oil by both benthic infauna and epifauna was discussed in Section 4.3.4.

4.4.5 Pathology and Histology

The literature reviewed during preparation of this report did not describe any histological or pathological effects of petroleum hydrocarbons exposure on cold-water marine epibenthic invertebrates. However, this is undoubtedly in part a reflection of the difficulties associated with examination of morphological changes in the tissues of invertebrates (particularly crustaceans). Pathological and histological effects of oil exposure may have occurred in some experimental studies and actual spills, and simply have not been detected (or examined) by researchers.

4.4.6 Behaviour

Several investigators have examined the behavioural responses of arctic epibenthic crustaceans to oil. In general, amphipods appear to be capable of detecting and avoiding hydrocarbons, whereas isopods have not exhibited consistent reactions when exposed to oil-contaminated food and sediments. In a laboratory study, the amphipods Onisimus affinis and Gammarus oceanicus generally avoided oil masses and showed a preference for unoiled over oiled food, while the isopod Mesidotea (Saduria) entomon did not avoid oil masses and readily consumed both oiled and unoiled food (Percy and Mullin 1975; Percy 1976). O. affinis showed a preference for clean rather than oil-contaminated sediments (0.05 mL oil: 15 g dry sediment), while the amphipod Corophium clarencense and the isopods M. entomon and M. sibirica did not react to the presence of oil in sediments (Percy 1977). Atlas et al. (1978) also reported that amphipods (but not isopods) avoided oil-contaminated sediments (200 mL oil: 1250 cm^3 sediment) resulting from an experimental spill of Prudhoe Bay crude. In addition, several authors have suggested that mortality of species that are indiscriminate particle feeders may result from ingestion of oil (Duval et al. 1981; Thomson and McLaren 1982).

Petroleum hydrocarbons dissolved in water have a narcotizing effect that is particularly noticeable in highly mobile organisms such as amphipods. Wells and Percy (1985) reported that erratic behaviour was frequently associated with narcotization. Impaired equilibrium, loss of upright posture, uncoordinated movement and erratic swimming have been observed in amphipods exposed to WSFs of oil (50 to 2000 ppm oil added) (Percy and Mullin 1975; Busdosh et al. 1978). Busdosh (1981, cited in Wells and Percy 1985) also found that activity in Onisimus affinis declined during exposure to hydrocarbons, and that the degree of inhibition was correlated with the concentration of hydrocarbons. At least initially, the reduction in activitywas a result of less time spent in spontaneous movement rather than a slowing of movement.

Documented behavioural responses of echinoderms to oil dispersions have included: retraction of tube feet which can result in loss of adherence to the substrate (North et al. 1965), decreased responsiveness to chemicals released by prey and reduced rates of predation (Rice et al. 1983), and interference with chemosensory defensive behaviour (Johnson 1977). An experimental oil slick during the BIOS program did not result in any observable (diver survey) behavioural responses by echinoderms (Cross and Thomson 1982), although it is important to note that elevated oil concentrations in the water column were restricted to approximately one meter below the surface.

4.4.7 Habitat Loss and Disturbance

As noted earlier, the potential for oil stranded in intertidal areas to be a chronic source of contamination of shallow sublittoral habitats makes the species that occur in these habitats particularly vulnerable to oil exposure. However, since many epibenthic species are mobile, they may be able to emigrate from contaminated to uncontaminated habitats. Amphipods may do this more frequently than isopods because they have been shown to avoid oil. The response of echinoderms to oil is not well documented, although affected individuals may consume less food because these species locate their prey by chemoreception.

4.4.8. Indirect Effects

Indirect effects of oil spills on benthic epifauna could result from decreased abundance of prey organisms. For example, the omnivorous sea urchin may be affected by a decline in abundance of macro-algae, while starfish could be indirectly affected by any large-scale decline in the abundance or availability of bivalves. The opposite effect may occur initially following an oil spill when starfish could opportunistically feed on any narcotized clams that are present on the sediment surface. Although both of the above first order declines in abundance are possible following oil spills (Sections 4.2 and 4.3), it is unlikely that second-order effects on predators would be significant. In the case of small spills, mobile organisms could move to uncontaminated areas, while in the case of larger spills, direct lethal and sublethal effects on epibenthic organisms would likely be of greater overall significance than any indirect effects.

4.4.9 Countermeasures

Chemical dispersants are relatively non-toxic in their free (unmixed) form, but because they increase the concentration of oil in the upper portion of the water column, there is concern that chemically dispersed oil may contact and affect benthic invertebrates that might otherwise be unaffected by a naturally dispersed surface oil slick. Foy (1982) found that the addition of Corexit 9527 to Prudhoe Bay crude oil increased the hydrocarbon concentration in the water column by as much as seven times when measured concentrations of chemically dispersed oil were compared to hydrocarbon concentrations generated by the same volume of oil dispersed by mechanical means. Gas chromatographic analysis of these physically and chemically prepared oil dispersions also indicated that a greater proportion of particulate oil and lower proportion of water-soluble aromatics were characteristics of the chemically dispersed oil (Foy 1982). The lower proportion of aromatic hydrocarbons in the chemically dispersed oil was cited as the reason for relatively lower toxicities of oil-Corexit-water mixtures than oil-water mixtures to the seven arctic amphipods and one copepod tested (96-h LC_{50} 32 to 73 ppm Prudhoe Bay crude oil dispersed mechanically compared to 45 to 196 ppm Prudhoe Bay crude oil dispersed with Corexit, Foy 1978).

Sublethal effects of chemically dispersed oil have been examined and compared to the effects of physically dispersed oil in some laboratory investigations. Duval et al. (1982) examined the sublethal effects of chemically and physically dispersed crude oil on the isopod Gnorimosphaeroma in the laboratory and found that chemically dispersed oil caused more pronounced sublethal physiological and behavioural effects than physically dispersed oil. Increasing exposure duration and chemically dispersed oil concentrations caused sublethal effects to increase from only uptake of napthalenes (1.1 mg L^{-1}; 24 h), to decreased assimilation, increased respiration, reducted moulting frequency and a decrease in mating behaviour (6.6 mg L^{-1}; 48 h [Duval et al. 1982]). At the highest dispersed oil concentration tested (6.6 mg L^{-1}), normal pairing activity was virtually eliminated and males lost their diel rhythm in external pigmentation. During an 8-wk recovery period following a 24-h exposure to 18.0 mg L^{-1} (WSF) of chemically dispered crude oil. test organisms recovered in terms of normal respiration and assimilation rates and fully depurated accumulated napthalenes during the same period (Duval et al. 1982). These experimental conditions did not result in adverse effects on growth or the number of emergent juveniles (Duval et al. 1982). The more pronounced sublethal effects observed following dispersed oil treatments were related to the greater opportunity for dissolution of water-soluble constituents of the crude oil and/or contact of organisms with individual oil particles. Nevertheless, Duval et al. (1982) concluded that short-term exposures to relatively low concentrations of dispersed oil would not necessarily affect the growth and survival of isopods.

Exposure to chemically dispersed oil in field and laboratory experiments during the BIOS program resulted in immediate but short-term effects on some epifauna. Responses of sea urchins (Strongylocentrotus droebachiensis) during laboratory exposures to dispersed oil concentrations of greater than 0.5 ppm included: debilitation of tube feet that caused loss of covering material and ability of organisms to adhere to the substrate, apparent impairment of the hydrostatic system resulting in flaccid peristomal membrane and periproct regions, narcotization resulting in loss of response to mechanical stimuli, uptake of hydrocarbons, and apparent neuromuscular effects resulting in loss of spine rigidity (Engelhardt et al. 1983). A dose-response relationship was established for the observed behavioural responses (Mageau and Engelhardt 1984) and recovery of most urchins occurred during a 7-day clearance period in clean water. The lethal concentration of dispersed oil for urchins in these investigations was calculated to be approximately 500 ppm. Behavioural responses of sea urchins observed in the field on the two days following the chemically dispersed oil release were similar to those documented during laboratory exposures, and included loss of response to mechanical stimuli, loss of adherence to the substrate and narcotization (Cross et al. 1984). Consistent results regarding the uptake of hydrocarbons by sea urchins have not been obtained during laboratory and field studies. Mageau and Engelhardt (1984) found that urchins exposed to 500 ppb dispersed oil in the laboratory accumulated only trace amounts of hydrocarbons, while sea urchins sampled from the field in areas exposed to dispersed oil had accumulated moderate amounts of oil. In addition, sea urchins exposed to chemically dispersed oil and physically dispersed oil accumulated higher concentrations of high molecular weight hydrocarbons when exposed to oil alone than when exposed to dispersed oil (Engelhardt and Norstrom 1982). No mortality of urchins was reported during field sampling and short-term (2 week) reductions in the density of urchins were assumed to be due to emigration of individuals from affected areas to deeper unaffected locations. No long-term changes in the density of 11 taxa of epibenthic crustaceans and 2 taxa of echinoderms resulted from the BIOS dispersed oil release (Cross et al. 1984).

4.4.10 Vulnerability

As stated in previous sections (4.1; 4.3.10), shallow benthic habitats are extremely susceptible to contamination by hydrocarbons following oil spills. Although some epibenthic fauna are mobile and may migrate to uncontaminated areas, others may suffer lethal or sublethal effects from hydrocarbon exposure. While displacement of individuals may be a temporary effect, especially if contamination is light and only a relatively small area is affected, sublethal effects of oil exposure could persist for longer periods. The magnitude of sublethal effects and the number of individuals in various populations that may be affected will depend on specific circumstances, such as the amount and physico-chemical properties of hydrocarbons entering the benthic environment. The seasonality is also an important factor determining the vulnerability of this group of invertebrates to oil spills, since larvae and moulting individuals appear to be more sensitive to on epibenthic invertebrates would be minimized in situations where only limited quantities of oil are physically (or chemically) dispersed in the water column, or reach benthic habitats because of direct sinking, deposition in zooplankton feacal pellets or transport from adjacent areas.

4.4.11 Recovery

Recovery of benthic epifauna following oil spill events would be expected to be somewhat faster than that of infauna (Section 4.3.11) due to the mobility of many species populations that comprise this community. Physical and biological processes important in the dissipation of oil from subtidal sediments would also play an important role in determining the period required for recovery. Lightly-contaminated areas may be rapidly recolonized by mobile taxa that do not avoid low concentrations of oil, as well as by those species attached to oil. On the other hand, recovery could require considerable periods if contamination of bottom sediments was extensive or if larval or moulting stages of epifauna were affected.

SECTION 4.5

MICRO-ORGANISMS

Dave Green and Blair Humphrey

4.5.1 Introduction

Marine micro-organisms are not only affected by petroleum hydrocarbons in the sediments and water column, but also have an important role in the long-term degradation of oil in marine environments. The following sections review available information on the uptake and metabolism of certain hydrocarbons by oil degrading bacteria, as well as the effects of hydrocarbons on populations and the physiology of micro-organisms. Emphasis has been placed on these processes in the cold marine environments that are the focus of this review.

Interactions between micro-organisms and oil can occur in all marine environments (subtidal and intertidal sediments, water column, within and on the surface of sea ice). However, since the majority of research has been directed at microbial populations and oil degradation in seafloor sediments, all information relevant to oil-microbe interactions is discussed in this chapter.

4.5.2 Oil Degrading Bacteria

Oil degradation by micro-organisms occurs within the water column, in sediments and on the surface of particles and masses. In spite of continuous major inputs, the levels of petroleum hydrocarbons detectable in the world's oceans are low, probably because of slow but ubiquitous degradation of oil by micro-organisms. Some oil-degrading bacteria (ODB) are found in all marine environments, although larger populations have been documented near sources of both natural (e.g. seepages at the seafloor) and unnatural (e.g. near sites of runoff from industrial centres) chronic petroleum inputs (Bartha and Atlas 1977, and references cited therein). The potential for oil degradation exists even in relatively pristine arctic regions. Populations of ODB have been found in Cook Inlet (Roubal and Atlas 1979), the Beaufort Sea (Bunch and Harland 1976), and in the eastern Arctic (Bunch 1979; Bunch et al. 1983). The proportion of ODB in arctic water bacterial communities varies from 0.01-1.6 percent in the Beaufort Sea to 10 percent in Cook Inlet. ODB have also been found in benthic and intertidal sediments. Many different bacterial genera have been shown to degrade oil (Bartha and Atlas 1977), although the most prevalent taxa are *Pseudomonas*, *Achromobacter*, *Arthrobacter*, *Micrococcus*, *Nocardia*, *Vibrio*, *Acinetobacter*, *Brevibacterium*, *Corynebacterium* and *Flavobacterium*.

Some marine fungi are also involved in oil degradation. For example, Ahearn et al. (1971, cited in Karrick 1977) found yeasts that were capable of growing in water containing up to 4 percent oil. These yeasts were present on the surface of single oil droplets. They developed mycelia that penetrated the oil and eventually developed into a fungal colony enclosing several oil droplets.

4.5.3 Population Effects

Chronic exposure to oil in the water column and in sediments stimulates the growth of oil-degrading bacteria. The research of Atlas (1978a) indicated that different oils and temperatures may encourage proliferation of different genera. Areas influenced by long-term exposure to oil have a lower diversity of genera and a higher proportion of ODB than pristine areas (Atlas 1978a). As a result, the presence of relatively large populations of ODB in an area can be used to verify the existence of chronic oil inputs. In temperate areas with oil-contaminated sediments, up to 80 percent of the bacterial populations have been shown to be ODB, whereas communities in uncontaminated areas have only 0.01 percent ODB (Azoulay et al. 1983). Large increases in the abundance of ODB have been observed in oiled intertidal sediments (Eimhjellen and Josefson 1984). Even areas only peripherally affected by oil can show an increase in ODB that is not evident in adjacent offshore benthic sediments, presumably due to the greater availability of oxygen in intertidal sediments. On the other hand, research by Bunch et al. (1983) suggested that short-term (acute) exposure to high levels of oil in the water column does not have an effect on the number of total viable heterotrophs or the relative abundance of ODB in the bacterial community (Bunch et al. 1983). Some microbial populations decline in the presence of oil, especially if no induction of oil-degrading enzymes by chronic hydrocarbon exposure has occurred (Atlas and Bartha 1973a).

4.5.4 Behavioural Effects

The only observed behavioural effect of oil on bacteria has been the loss of chemotaxis in some populations. However, this effect is reversible and does not result in loss of motility (Mitchell et al. 1972).

4.5.5 Sublethal Effects

In laboratory and field experiments, populations of micro-organisms in water and sediments have shown decreases in glucose and glutamate uptake in the presence of oil, with a corresponding increase in percent glutamate incorporated in the cell or evolved as CO_2 (Griffiths et al. 1981a,b). These authors suggested that biosynthetic pathways were most affected by hydrocarbon exposure. The effects of hydrocarbons on biosynthesis by micro-organisms have been more noticeable in clean water than in water which has been contaminated with oil. Bunch (1983) did not observe any effects of oil on micro-organisms in water, as indicated by changes in glutamic acid uptake, unless a dispersant (Corexit 9527) was added to the test media.

4.5.6 Morphology

Intracytoplasmic inclusions have been observed in micro-organisms exposed to oil, and may greatly increase cell size (Karrick 1977). Devereux and Sizemore (1982) associated the presence of plasmids with bacterial communities that have a high proportion of ODB, and suggested that plasmids may assume an important role in oil degradation.

4.5.7 Hydrocarbon Uptake, Transformation and Depuration

Several components of oil are degraded by micro-organisms. Aerobic degradation of hydrocarbons in the water column and at the sediment surface is common. There is also some evidence for anaerobic degradation in subsurface or deep ocean sediments, although both the amount and rate of degradation are low (Ward et al. 1980). In Arctic waters, aerobic degradation of alkanes and aromatics proceeds at equal rates (Atlas 1978a), but in most laboratory and field studies, alkanes have been degraded more rapidly than aromatics.

The following summary of the metabolic processes of bacterial biodegradation is based on a review by Bartha and Atlas (1977). Straight-chain alkanes are most rapidly degraded, typically by beta-oxidation, from alkane to alcohol to aldehyde to acid to shorter acid. Hydrocarbons with chain lengths of 10 to 25 carbons are rapidly utilized during aerobic degradation. Shorter paraffins are toxic to micro-organisms, while longer paraffins are too insoluble to be absorbed and metabolized. Branched chain alkanes are utilized much more slowly than straight-chain alkanes, since only primary carbons can form an acid; secondary carbons are oxidized via a diacid, while tertiary and quaternary carbons are rarely degraded. Cyclic alkanes are not completely degraded because the rings remain stable.

Aromatic components are normally oxidized to diols via mixed function oxidases (Varnasi and Malin 1977); the ring is then cleaved and the subsequent diacid further degraded. Substituted aromatics are degraded by removal of the side chain and then opening of the rings. The cleaved rings enter the tricarboxylic acid system and are further degraded (Karrick 1977).

Some components in oil, in particular aromatics (Gibson 1977), are degraded by cometabolism. During this process, substrates which result in products that are unsuitable for sustaining growth are metabolized at the same time as substrates that give useful degradation products. Cometabolism relies on the induction of enzymes that not only utilize growth supporting substrates, but also their less useful analogues (Karrick 1977). When the usable substrates are depleted, the metabolism of other substrates also normally stops.

A typical biodegradation sequence involves the rapid loss of paraffins from C10 to C25, leaving isoprenoids such as pristane. The latter hydrocarbons are also degraded once the paraffins are degraded. This leaves an unresolved complex mixture containing high molecular weight cyclic and aromatic compounds.

The products of aerobic microbiological degradation are usually alcohols, aldehydes and acids, although some hydroperoxides, ketones and esters are also produced (Atlas 1978b). These degradation products can be more toxic than their hydrocarbon precursors. In particular, the alcohols of the C6 to C9 paraffins are more toxic than the parent compounds, and some fatty acids and many aromatic oxidation products are more toxic than precursor hydrocarbons (Lui 1973, Atlas and Bartha 1973b, both cited in Bartha and Atlas 1977). Karrick (1977) also reported that the metabolism of oil by bacteria may change lipid type from glycerides to waxes.

The anaerobic degradation of oil has not been well documented, but appears to involve the use of sulfate as a final electron donor. Hydrocarbons do not appear to have an effect on anaerobic bacteria (Winfrey and Ward 1981).

Factors limiting the microbial degradation of oil include temperature, oxygen content and availability, and nutrient concentrations. Sea water and sediment temperatures in cold marine environments may vary from -1 °C to 4 °C. Although degradation occurs at these low temperatures, it is generally slower than in temperate waters. Oxygen availability is the most critical factor in the biodegradation of oil because most of the documented metabolic processes involve oxidation. Oxygen is generally present at high concentrations in surface sea water, but the oxygen level is often too low to support active aerobic degradation in very deep water and in marine sediments. At some times of the year, such as after the spring phytoplankton bloom, nutrient levels may also be low within the upper portion of the water column. Under such circumstances, bacterial degradation of oil may continue, but often results in decreased reproductive rates and the production of a flocculant material known as "marine snow" (Green et al. 1982). On the other hand, oxygen and nutrient levels in shoreline areas are generally high, and these habitats therefore frequently support abundant bacterial populations.

Calder and Boehm (1981) suggested that microbial degradation is a major pathway for removal of oil from surface sediments. Similarly, Atlas and Bronner (1981) estimated that microbial degradation could remove 0.5 kg hectare^{-1}day^{-1} of petroleum hydrocarbons from intertidal sediments.

There has been limited investigation of bacterial communities associated with ice, although Atlas et al. (1978) documented the presence of ODB both on and under sea ice in the Canadian Arctic. During experimental exposures to oil, the proportion of ODB in the bacterial community increased 3 to 5 times, and was accompanied by a corresponding increase in the percentage of heterotrophic organisms.

4.5.8 Effects of Countermeasures

Oil spill countermeasures such as the use of sinking agents and chemical dispersants may increase the amount of oil reaching benthic environments. For example, the application of sinking agents such as chalk or charcoal may lead to deposition of relatively unweathered oil on the seafloor. Similarly, the use of dispersants in inshore or intertidal areas could lead to accumulation of oil in adjacent subtidal environments. This could be of concern in arctic environments where the biomass of benthic fauna is higher in subtidal areas than in intertidal habitats.

Few investigations have examined the effects of chemical dispersants and dispersed oil on bacterial populations. During the BIOS Project, a transient increase in glutamic acid uptake was observed in the presence of large quantities of dispersed oil in the water column, although numbers of bacteria were not affected (Bunch and Cartier 1984). Bunch (1983) reported that relatively high concentrations (0.01 percent) of Corexit 9527 inhibited hexadecane mineralization by bacteria during laboratory studies. This chemical dispersant has also been shown to inhibit alkane degradation in the absence of nitrogen or phosphorus (Foght and Weslake 1982). Other studies have shown that chemically dispersed oil increases oxygen demand, probably due to the larger surface area of oil that is available for bacterial attack and the subsequent high activity of bacterial populations (Karrick 1977). The application of dispersants to floating oil slicks often results in oil dispersions characterized by a much smaller particle size than when oil is dispersed in the upper water column through natural oceanographic mixing processes. This increases the total surface area of the oil and ,therefore, the amount of substrate available for settlement and growth of bacteria. The resultant proliferation of bacteria can then deplete available nutrients from adjacent waters and induce the production of "marine snow" and inhibit further reproduction.

Eimhjellen et al. (1982) suggested that fertilization of water surface or intertidal areas contaminated with oil may be a potential countermeasure because it can lead to significant increases in biodegradation rates. Eimjellen and Josephson (1984) showed that fertilization of intertidal areas contaminated with oil could cause a 10 to 100 fold increase in ODB populations.

At one time, the "seeding" of oil-contaminated areas with bacterial cultures was proposed as a countermeasure, although it is now evident that most habitats already support some ODB populations. Consequently, this countermeasure would probably be unnecessary in most instances.

CHAPTER 5

OIL ASSOCIATED WITH ICE

The biological effects of oil spills in arctic or subarctic regions during periods of complete or broken ice cover have been identified as a significant area of concern by a large number of scientists. The increasing emphasis on oil exploration in these regions and the potential for hydrocarbon production and transportation in many parts of the Canadian Arctic (e.g. Beaufort Sea and Parry Channel) has provided the impetus for several reviews of the available literature on the impacts of oil spills in arctic regions. The most notable of these reviews is the two-volume set. "Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms" (D.C. Malins, ed. 1977, Academic Press). At the time of publication, contributing authors had to rely extensively on information collected during studies with temperate and even tropical species, and observations from spills that occurred outside arctic and subarctic regions. Since then, however, a large body of knowledge relevant to arctic species and environments has been assembled through the various studies conducted during the 1980-1984 Baffin Island Oil Spill Project. The results of various investigators that participated in this program form the basis of much of this chapter.

Subsequent sections in this chapter review the effects of oil on organisms that occur within or under the ice (epontic biota), and provide a brief over-view of the physical and chemical fate of oil that contacts ice. However, it should be emphasized that a number of organisms discussed in other sections of this report occur in areas of complete or broken ice cover at some times of the year and could be affected by oil in these habitats. The most notable of the groups that may be associated with the ice cover at certain times of year are some seabirds such as eiders (Section 2.2), pinnipeds such as ringed and bearded seals (Section 2.3), polar bears (Section 6.5) and several fishes (e.g. Arctic cod; Section 3.4). In addition, oil encapsulated in ice and released during the spring melt could affect phytoplankton, zooplankton and the larval stages of benthic and intertidal fauna and some species of fishes.

SECTION 5.1

OVERVIEW OF PHYSICAL AND CHEMICAL PROCESSES

Dave Green and Blair Humphrey

Oil spilled or transported under first-year ice tends to collect in depressions in the rough lower ice surface. The oil which gathers under growing ice apparently blocks the brine drainage process (Mackay 1984). This oil is rapidly encapsulated as a lens in the ice sheet, where it remains immobile and inert until spring. Very little weathering other than some dissolution and loss of volatiles occurs during this period because of the low temperatures in and under the ice. As the ice melts, oil migrates up the brine channels to the ice surface, and reaches the ice surface well before an appreciable fraction of the ice has melted (Mackay 1984). Until breakup, oil contaminates primarily the upper surface of the ice. As temperatures increase, the oil on the ice surface collects in melt pools; the oiled surface is preferentially melted due to its decreased albedo. This oil is then herded by winds and may quickly weather and form mousse. As breakup progresses, the weathered oil is released to the water surface. These processes are described in detail in MacKay (1984) and illustrated in Figure 5.1-1 which is from this source.

Oil present under ice in the spring, or spilled in areas of ice cover during the period of ice melt, may remain in pockets or coat the rough underside of the ice until breakup. Some weathering of this oil occurs, and mousse formation is encouraged by movement of the oil against the ice undersurface.

The behaviour of oil under multi-year ice and in areas of discontinuous ice cover has been less studied, but it is expected to be substantially different than the processes described above. In multi-year ice, there are no brine channels to facilitate oil migration to the surface.

Nevertheless, Mackay (1984) reports that oil trapped under this ice type may reach the surface by movement through cracks and by ice ablation. In the case of broken ice, oil naturally migrates to the air-water interface and may become associated with slush ice in areas of open water (Mackay 1984).

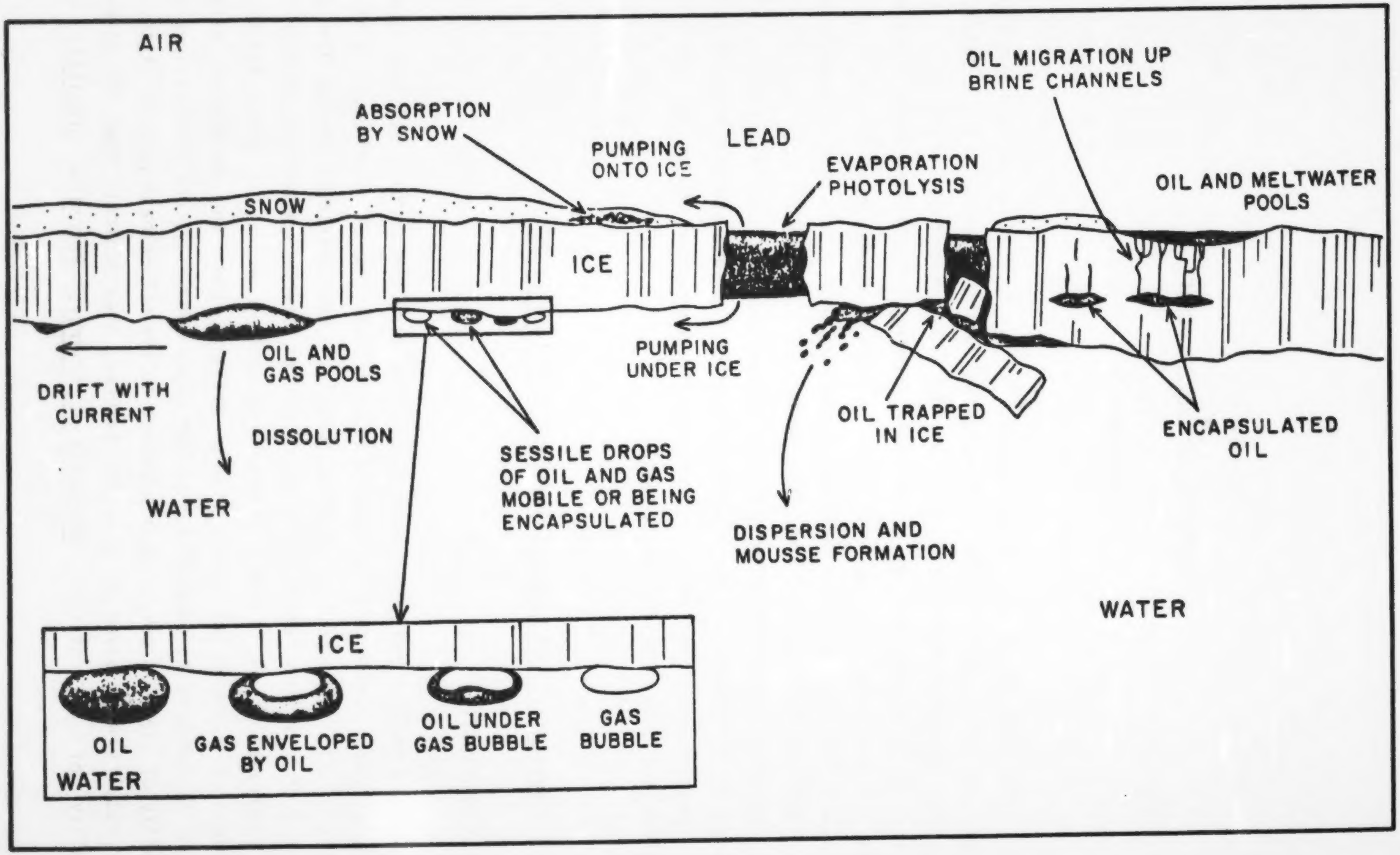

Figure 5.1-1 Summary of Interactions of Oil With Ice (Source: Mackay 1984)

SECTION 5.2

EPONTIC FLORA AND ICE ALGAE

Linda Martin

The effects of oil on epontic flora (ice bottom algae) and ice algae (ice column algae) were recently investigated during experimental oil releases in McKinley Bay (Beaufort Sea) and during the BIOS project at Cape Hatt (Baffin Island). Prior to the initiation of these studies in 1979, the effects of oil spills or under-ice oil well blowouts on these communities could only be postulated from data on planktonic species that also occur in the epontic and ice column communities, and from information on the probable behaviour of oil under the ice. Since the effects of oil on some epontic species (particularly diatoms) may be similar to those documented with the same species in planktonic algal communities (Section 3.2), cross-references to earlier sections are provided throughout the following text.

5.2.1 Sensitivity of Epontic Flora and Ice Algae to Oil

5.2.1.1 Mortality

Ice-bottom and ice-column algae that directly contact oil are likely to be killed. Crude oil is toxic to some species of planktonic diatoms (Section 3.2), and species-specific mortality was indicated during the one investigation of the lethal effects of oil on diatom communities within sea ice. Acreman et al. (1980) examined ice-column algae in cores following experimental releases of crude oil in McKinley Bay, and documented slight changes in the species composition of the algal community, and species-specific mortality of algae in cores taken adjacent to the oil release points. Although their results were based on very few samples, they indicate that the percent mortality of the diatoms *Nitzschia cylindrus*, *Porosira glacialis* and

Coscinodiscus lacustris was greater than the average mortality for all diatoms. Moreover, control samples contained higher numbers of cells and were dominated by pennate diatoms, while centric diatom species (e.g. Chaetoceros fragilis) were more prevalent in ice column cores affected by oil but were present at lower densities. Acreman et al. (1980) suggested that the observed changes in species composition and abundance following experimental releases of crude oil were greater than those associated with natural variability.

5.2.1.2 Contamination and Physical Damage

Acreman et al. (1980) provide the only data describing contamination or physical damage to ice algae. They found abnormal dark particles and a clear substance that occupied a large percentage of certain diatom cells. Although these substances were not identified at the time, the investigators suggested that they were the result of oil contamination. However, subsequent investigations have shown that the dark particles were natural degradation products and that the clear substance was a storage product, and probably neither feature was due to exposure to the oil (J. Acreman, pers. comm.).

5.2.1.3 Photosynthesis and Production

During the BIOS program, the effects of oil on epontic algal productivity (mg C m^{-2} h^{-1}), density (cells m^{-2}), and chlorophyll a concentrations (mg m^{-2}) were examined utilizing both laboratory and in situ experimental techniques. Cross and Martin (1983) reported considerable spatial and temporal variability in the productivity and standing stock of epontic algae, but observed little evidence of adverse effects of oil on this community. In contrast to anticipated results, these researchers found that crude oil apparently stimulated productivity and increased the biomass of some algae. They reported that stimulation of algal production was not correlated with increases in light, and suggested that the oil treatments were responsible for the observed effects.

Cross (1982) found that an oil concentration of 10,000 mg L^{-1} inhibited photosynthesis of epontic algae, but observed no measurable effects at lower oil concentrations (10-300 mg L^{-1}). Van Baalen and O'Donnell (1984) also observed no change in the rate of photosynthesis of Bering Sea ice diatoms (Nitzschia sp.) after 3-h treatment with moderate concentrations of crude oil (625 mg L^{-1}) and fuel oil (250 mg L^{-1}). Hsiao et al. (1978) examined the effects of oil on photosynthesis of phytoplankton from the Beaufort Sea and found that higher oil concentrations inhibited primary production, while lower oil concentrations occasionally caused an increase in photosynthetic production. Although the mechanism for stimulation of algal photosynthesis by hydrocarbons is not known, Hsiao et al. (1978) suggested that the presence of growth-regulating compounds and/or trace metals in oil acted as micronutrients and were responsible for the observed trends.

5.2.1.4 Growth and Reproduction

Recent laboratory investigations of algal growth, using diatom cultures (Nitzschia sp. and Chaetoceros sp.) grown in the presence of crude oil and fuel oil, indicated that growth of these species was inhibited to a much greater degree at 0 °C than at 10 °C (Van Baalen and O'Donnell 1984). Although a metabolic basis for toxicity was not established, these researchers suggest that psychrophilic algae may be about five times more sensitive than mesophilic algae to petroleum pollution.

5.2.1.5 Uptake, Metabolism and Depuration of Hydrocarbons

Cerniglia et al. (1981, cited in Van Baalen and O'Donnell 1984) reported that aniline absorbed by blue-green algae is transformed to acetanilide, formanilide and p-aminophenol. Although no other studies of this type appear to have been directed at epontic algae, metabolism and depuration of hydrocarbons by phytoplankton has been documented (Section 3.2), and may occur in algal populations found within and under the ice.

5.2.1.6 Habitat Loss and Disturbance

There are two effects of oil accumulation on under-ice and ice-column habitats (1) physical coating of the lower surface of the ice, which would isolate ice-column algae from their nutrient source and also from grazers (Acreman et al. 1980), and (2) oil migration through the ice column, subsequently decreasing light transmittance to both algal communities. Decreased albedo of oiled ice also increases the melt rate of the ice sheet in spring and, therefore, could reduce the period of primary production by this community. In addition, in the spring, ice algae would be exposed to oil that migrates through brine channels. Acreman et al. (1980) reported reduced algal abundance and changes in algal species composition in an ice core containing oil, although virtually none of the above potential habitat-related effects of oil on epontic flora and ice algae have been investigated to date.

5.2.1.7 Effects of Countermeasures

The use of dispersants and oil solidifiers are the only countermeasures whose effects on epontic algae have been investigated. There is no information on the effects of oil ignition at the ice surface on biological communities within or on the lower surface of ice. The results of the BIOS studies indicate that a dispersed oil concentration of 10,000 mg L^{-1} (Corexit 9527 and Lagomedio crude) caused total suppression of algal productivity, whereas untreated oil only resulted in a reduction in productivity (Cross 1982). However, it is not clear if this greater apparent decrease in productivity at high dispersed oil concentrations was real or the result of different laboratory treatments (i.e., longer time in separatory funnel). There was some evidence of stimulation of photosynthesis at the lowest dispersed oil concentration (10 mg L^{-1}) used during these studies. Hsiao et al. (1978) also noted some increase in production in arctic phytoplankton exposed to dispersed oil.

During 1982, one of the BIOS studies examined the effects of crude oil mixed with three dispersants, solidified oil, and by itself on the standing stock and productivity of epontic algae. No immediate effects of any dispersed oil treatment (5.8 to 36.5 ppm) on the productivity or biomass of algae were detected, although some under-ice areas treated with oil alone and with solidified oil had a higher algal biomass (cells m^{-2}) and productivity (chl a mg m^{-2} and mg C m^{-2} h^{-1}) than measured in control areas (Cross and Martin 1983).

5.2.2 Vulnerability

Epontic algae would be most vulnerable to effects of spills or blowouts in areas where oil accumulated in depressions under the ice. Present information suggests that productivity of ice algae would not be adversely affected in areas that were lightly contaminated by oil, although other sublethal effects such as reduced growth might occur. The vulnerability of under-ice flora to the effects of oil would be greatest during the late spring and early summer when productivity is highest. Oil contacting the under-ice surface earlier in the year during growth of the ice sheet would be encapsulated, but could still interfere with light transmittance and reduce algal productivity during the spring bloom.

SECTION 5.3

EPONTIC FAUNA

Linda Martin

In situ studies of the effects of oil on epontic fauna were completed during the BIOS project (Cross 1982; Cross and Martin 1983). In addition, a number of researchers conducted laboratory investigations of the effects of oil on epontic invertebrate species that were collected from pelagic, littoral and sublittoral habitats (e.g. Percy 1976, 1977a,b; Percy and Mullin 1975, 1977; Busdosh and Atlas 1977; Foy 1978, 1979). These studies have provided important information on the relative sensitivities of some under-ice invertebrate species to oil.

5.3.1 Sensitivity of Epontic Fauna to Oil

5.3.1.1 Mortality

Oil accumulation under the ice will result in mortality of some epontic fauna. During *in situ* studies, Cross (1982) found that the degree of mortality of *Weyprechtia pinguis* was directly related to the extent of oil contamination, and that only very lightly contaminated amphipods survived oil exposure. In addition, immediate mortality increased with increasing oil concentrations (0 to 400 mg L^{-1}). Although the duration of oil exposure did not appear to be a factor affecting mortality of amphipods during the investigation, delayed mortality (over a 4-d post-exposure period) was higher following a 3-h oil exposure, while immediate mortality was higher following a 15-h oil exposure (Cross 1982). Fifteen-hour exposures to oil concentrations of 400 mg L^{-1} resulted in >96 percent mortality of test organisms; exposure at the lowest oil concentration and shortest time (30 mg L^{-1}; 3 h) caused 68 percent mortality. Foy (1978, 1979) found a slightly higher oil tolerance in

Onisimus (96-h LC_{50} = 49 mg L^{-1}), and determined that juveniles of this species were less sensitive to oil than adults (96-h LC_{50} of 68 mg L^{-1} compared to 96-h LC_{50} of 49 mg L^{-1}). Even brief exposure to relatively low oil concentrations will result in mortality of some species of epontic fauna.

Cross and Martin (1983) reported that the mortality of 0+ year class Gammarus setosus was greater than mortality of older individuals following oil slick contact in intertidal habitats at Cape Hatt. The observed oil effects persisted in the epontic community during the following spring, when a marked reduction in the number and size of newly-released juveniles and one-year-old individuals was noted. Although this cohort was almost absent from the under-ice surface, post-spill sampling indicated that this year class was present in the intertidal habitat one-year following the oil release.

5.3.1.2 Growth and Reproduction

The effects of oil on growth and reproduction of epontic fauna were examined during the 1982 BIOS studies. Cross and Martin (1983) determined the density, biomass and length frequency distributions of major epontic, intertidal and sublittoral amphipod species in control, oil-contaminated and dispersed oil-contaminated areas in the year following the August 1981 experimental oil releases. Since most species in the epontic community originated in littoral or sublittoral habitats (Cross and Martin 1983), it was important to simultaneously examine the effects of oil in each of these areas. The researchers found that in the spring following the oil release, the number of juvenile (newly released) and one-year-old Gammarus setosus on the under-ice surface was lower than in the previous year, and suggested that this reduction was due to immediate mortality of these year classes in intertidal populations of Gammarus after direct oil contact and due to delayed release of juveniles. This delay was expected to be responsible for reduced body size of the 0+ year class amphipods in the intertidal habitat during August and September. The median size of this 0+ year class was 4 mm in contrast to 5 mm in the previous year.

5.3.1.3 Physiology and Metabolism

Some data on the sublethal metabolic and physiological effects of oil are available for Onisimus as a result of the Beaufort Sea Project studies (Percy and Mullin 1975). These investigators found that oil-in-seawater dispersions at low concentrations depressed respiration rates, while increasingly higher oil concentrations reversed the inhibition. At these higher concentrations, the respiration rate of Onisimus either returned to the control level or increased to substantially higher rates. Percy and Mullin (1975) suggested that the observed physiological responses of this species to oil are quite complex, and that a variety of endogenous and exogenous factors may have a pronounced influence on the magnitude and direction of changes in metabolic rates.

5.3.1.4 Uptake, Metabolism and Depuration of Hydrocarbons

The uptake and transformation of petroleum hydrocarbons by invertebrates has not been directly examined in epontic habitats *per se*. However, as indicated earlier, many of the species associated with the lower surface of ice are also found in benthic habitats in other areas and seasons. Consequently, it is assumed that information collected on epibenthic invertebrates and described elsewhere (Section 4.4) is at least partly relevant to epontic fauna. In addition, the fate of hydrocarbons accumulated by under-ice fauna may be comparable to that of hydrocarbons accumulated by crustaceans in intertidal habitats (Section 6.3).

5.3.1.5 Behaviour

Abnormal behaviour patterns of amphipods were found in both oiled and control chambers positioned under the sea ice during BIOS studies; Cross (1982) suggested that crowding in the chambers, rather than oil, was the cause of the observed behavioural responses. Amphipods did not avoid oil injected into under-ice chambers, and direct contact of amphipods with oil droplets and settling of individuals on the contaminated under-ice surface was observed. This lack of avoidance of oil by amphipods has also been documented by other investigators (Busdosh and Atlas 1977). On the other hand, Percy (1976) found that Onisimus avoided oil-contaminated food, although pre-exposure to oil or weathered oil reduced avoidance responses in this species. Busdosh and Atlas (1977) and Cross (1982) suggested that the apparent lack of response of arctic amphipods to oil would substantially increase their susceptibility to oil spills.

Cross (1982) reported that impaired locomotory ability caused 80 to 100 percent of the test organisms to remain on the bottom of chambers following exposure to 130 and 400 mg L^{-1} of oil. Laboratory studies conducted by Percy and Mullin (1977) also indicated that locomotory activity of Onisimus decreased with increasing oil concentrations, and that recovery of locomotory ability did not occur during a 24-h post-exposure period in clean seawater. These researchers found that physical fouling of appendages was not responsible for the reduction in locomotory activity, and suggested that a neuralmuscular disorder was responsible for the observed effects.

5.3.1.6 Habitat Loss and Disturbance

It is unlikely that under-ice surfaces coated with oil or dispersed oil would be inhabitable by epontic fauna. Not only would algal and meiofaunal food sources be eliminated or drastically reduced in abundance in these areas, but individuals contacting oil probably would not survive if they were extensively contaminated. On the other hand, epontic habitat would be available to algal and faunal colonization once any subsurface oil was encapsulated in the growing ice sheet. Oil entering areas of open water and broken ice from surface melt pools during spring could re-contaminate epontic habitats, although this is not very likely because such oil would be sufficiently weathered and ,therefore, would tend to sink away from the lower surface of the ice.

5.3.1.7 Effects of Countermeasures

The 1981 BIOS study results indicated that mortality of Weyprechtia exposed to chemically-dispersed oil (30, 130 and 400 ppm in controlled under-ice exposures) was 50 percent less than mortality caused by exposure to untreated oil (Cross 1982). In laboratory studies conducted by Foy (1978, 1979), mortality was higher in test organisms exposed to oil-Corexit-water mixtures than in oil-water mixtures of the same oil concentration. Foy (1979) suggested that the higher mortality observed in oil-dispersant tests was a result of higher hydrocarbon concentrations in the water containing chemically-dispersed oil. Some population level effects were attributed to dispersed oil releases during the 1982 BIOS investigations of epontic amphipods and meiofauna (Cross and Martin 1983). In areas affected by dispersed oil, these investigators found: (1) inhibition of recruitment of Onisimus; (2) size-specific mortality of Ischyrocerus; (3) emigration or mortality of older (1 and 2+ year classes) Gammarus; and (4) reduced densities of copepods and polychaetes in the meiofaunal component of the epontic community. The existing data do not provide clear evidence either in favor of or against the use of dispersants if oil threatens epontic habitats.

5.3.1.8 Indirect Effects

Indirect effects of oil spills on epontic communities could result if chemically dispersed oil causes mortality of epibenthic fauna in littoral and sublittoral habitats, and thereby reduces colonization of under-ice surfaces. Both these habitats are very susceptible to oil accumulation and since crustaceans are also relatively sensitive to oil, any significant losses of these fauna in benthic habitats could subsequently affect recruitment to epontic habitats. Decreased abundance of epontic algae and meiofauna may also indirectly affect epontic fauna due to localized reductions in food availability. However, neither of these indirect effects would be expected to be significant in terms of regional populations of epontic fauna due to the opportunities for recruitment of individuals from adjacent uncontaminated areas and the discontinuous nature of oil contamination in both sublittoral and epontic habitats.

5.3.2 Vulnerability

Epontic fauna are highly vulnerable to effects of oil because: (1) oil will accumulate in under-ice habitats; (2) oil under and encapsulated in ice will reduce the abundance of algal and meiofaunal food sources; and (3) oil may also decrease the abundance of the littoral and sublittoral populations that form the basis of this community.

CHAPTER 6

OIL ON SHORELINES

This chapter provides an overview of the behaviour and fate of oil stranded in intertidal and backshore habitats, and summarizes available information regarding the biological effects of oil on flora and fauna that occupy this habitat for a major portion of their life history. The following marine resources are discussed in this chapter: (1) intertidal algae and marsh grasses; (2) intertidal fauna; (3) marine-associated mammals (e.g. sea otters and polar bears); and (4) shorebirds and wading birds. The gametophyte stages of non-vascular plants and the larvae of most intertidal invertebrates may be exposed to oil in this habitat at certain times of the year. Similarly, polar bears and sea otters are often present in waters away from shoreline or ice edge habitats and may contact oil on the water surface or within the water column. Several species of birds discussed earlier in Section 2.2 stage or nest in shoreline and backshore habitats and may contact oil in these areas.

SECTION 6.1

OVERVIEW OF PHYSICAL AND CHEMICAL PROCESSES

Dave Green and Blair Humphrey

The behaviour and fate of oil reaching shorelines is affected by a large number of factors including: (1) season; (2) oil type and volume and the degree of its weathering before reaching shore; (3) the nature of the shoreline and the degree to which it is exposed to waves; (4) the nature of the intertidal zones it reaches; and (5) the countermeasures and oil recovery methods used following the spill. Each of these factors are discussed in detail by Owens (1984).

Fresh oil stranded on shorelines quickly undergoes some weathering and then resists further weathering. Initial losses occur largely through evaporation and dissolution; up to 40 percent of the total volume of some oils may be lost in these ways in the first few days after a spill. The remaining viscous oil mass contains the higher molecular weight hydrocarbons, including long-chain alkanes, cycloalkanes and polycyclic aromatics. Breakdown of this oil by oil-degrading bacteria is extremely slow and results in only a small decrease in volume. Microbial degradation of petroleum hydrocarbons was discussed earlier in Section 4.5.

Reworking of beach sediments by wave action and ice movement may bury stranded oil masses or transport them to higher areas in the intertidal zone. Some oil may also be transported from intertidal to subtidal habitats (Section 4.1). In some cases, the oil stranded in the upper intertidal zone may harden and form an asphalt-like substance that persists for a long time (Owens 1984). When the air temperature rises, the oil masses soften and ooze liquid hydrocarbons. This phenomenon was well documented following the METULA spill (Duval et al. 1981). Oil buried in intertidal sediments resists further weathering and may remain for extended periods.

The persistence of oil in intertidal habitats depends primarily on the substrate type and on the degree of shoreline exposure to wave action. Oil that does not penetrate beach sediments is generally removed more rapidly than oil within sand, gravel or cobble substrates. Exposure to wave and tidal action also influences the persistence of stranded oil. Oil on exposed high-energy shorelines is rapidly removed, often within a single season (Owens 1977), while oil on lower-energy shorelines can persist for several years (Owens 1984). A review of oil spill case histories (Duval et al. 1981) indicated that persistence of oil is shortest in rocky habitats exposed to considerable wave action, while oil in salt marshes and sheltered intertidal areas with sand substrates persists for the longest periods.

SECTION 6.2

INTERTIDAL ALGAE AND MARSH GRASSES

Linda Martin

Effects of oil spills on intertidal algae and marsh grasses have been frequently documented, largely because intertidal zones are often among the most visibly affected habitats following oil spills. Forty-one of the 100 marine oil spills reviewed by Duval et al. (1981) resulted in damage to or mortality of marsh vegetation or intertidal algae. Johnson (1977) discussed the results of laboratory studies completed prior to 1977 on the effects of oil on arctic and subarctic algae. These two major reviews summarize most of the available data regarding the effects of oil spills and petroleum hydrocarbons on plant communities.

6.2.1 Sensitivity to Oil

6.2.1.1 Mortality

Mortality of intertidal algae and marsh grasses following oil spills has resulted from: (1) coating and suffocation of plants; (2) direct toxic effects; (3) dislodging of plants due to the added weight of stranded oil; and (4) damage or removal during shoreline cleanup activities (see Section 6.2.1.7). In many cases, marsh grasses such as Spartina have been more severely damaged and required longer to recover from oil than intertidal algae (Duval et al. 1981). Oil penetration and retention in the sediments and its subsequent availability to rhizomes of plants has been found to be the cause of long-term toxic effects of oil on marsh grasses. Baker (1970) suggested that oil in the sediments prevented oxygen from reaching Spartina root systems. Thomas (1973) also reported toxic effects of oil on Spartina following the ARROW spill. Death occurred in the second winter after the spill, when aerial shoots were absent and rhizomes were exposed to highly anoxic sediments. Thomas (1973) suggested that in marshes with high natural sedimentation rates, the recovery rate of marsh grasses would be increased if some roots remained undamaged after the spill and subsequent cleanup, and were then able to penetrate newly deposited, uncontaminated sediments.

Mortality of all types of intertidal algae has generally been greatest under one or more of the following conditions: (1) the entire intertidal zone was coated with oil; (2) oil was stranded in upper zones over successive tidal cycles; and (3) oil retained in sediments was released over a long period of time following the spill. In cases where little oil has reached shoreline areas or where wave action rapidly removed the stranded oil, mortality of algal communities has been less pronounced (Duval et al. 1981). For example, Fucus sp. was unaffected by the ELENI V fuel oil spill in England and the TSESIS bunker spill in Sweden. Experimental oiling of Fucus spiralis on the English coast also had no observable adverse effects (Crothers 1983). Several authors have suggested that the mucilaginous coatings of some brown algae may prevent oil from adhering to their fronds, thereby reducing the effects of oil (e.g. Nelson-Smith 1970, Thomas 1973). Following the ARROW spill, Fucus vesiculosus and Ascophyllum nodosum were unaffected by the oil, due apparently to their mid-intertidal habitat and mucilaginous coating. In contrast, Fucus spiralis present in the upper intertidal suffered delayed mortality during the 10-month period following the ARROW incident (Thomas 1973). The specific circumstances surrounding each oil spill, and in some cases species- specific sensitivity to oil, determine the extent of mortality of intertidal algae.

6.2.1.2 Photosynthesis and Production

Johnson (1977) reported that the rate of photosynthesis by marine macro-algae could be stimulated or depressed by oil exposure depending on the species, amount of hydrocarbons present, and exposure conditions. For example, studies conducted by Yentsch et al. (1973, cited in Johnson 1977) indicated that net photosynthesis in the intertidal algae Polysiphonia lanosa, Porphyra *umbilicalis*, *P. minita*, *Spongomorpha arcta*, *Gigartina stellata* and *Fucus vesiculosus* was relatively unaffected by 24-h exposures to sea water extracts of various crude oils. Two- to 4-h treatments with 7 mg L^{-1} of crude oil did not affect photosynthesis in *Fucus distichus*, *Palmaria palmata*, *Halosaccion glandiforme*, *Costaria costata* and *Alaria tenuitolia* (Shiels et al. 1973, cited in Johnson 1977). However, photosynthesis in *Palmaria palmata* and *P. umbilicalis* was reduced during 24-h exposure to 100 percent and 50 percent crude oil concentrations, respectively. Inhibition of photosynthesis was also documented in *Cladophora*, *Ulva* and *Laminaria* spp. as a result of 2- to 4-h exposures to 7 mg L^{-1} crude oil (Shiels et al. 1973, cited in Johnson 1977). Reports of bleached intertidal algae following oil spills (Duval et al. 1981) indicate that photosynthesis has been inhibited, but the degree of this effect has not been documented in the case history literature. On the other hand, photosynthetic rates in some other genera (*Ulva*, *Alaria* and *Costaria*) were increased by exposure to low concentrations of oil extracts (0.7 mg L^{-1} for the former two species and 0.007 mg L^{-1} for the latter species). Intertidal algae are exposed to dissolved and dispersed oil in the water column, but exposure to surface oil is more critical. Schramm (1972, cited in Johnson 1977) reported that thin films (0.0001 to 0.1 mm) of several crude oils damaged *Fucus vesiculosus*, *Laminaria digitata*, *Porphyra umbilicalis* and *Enteromorpha sp*. by reducing CO_2 exchange and photosynthesis.

6.2.1.3 Growth and Reproduction

A wide range of effects of oil on growth and reproduction in intertidal algal communities have been observed following spills. These include: (1) reduced growth and reproduction in marsh grasses; (2) increased seed and flower production in *Spartina*; and (3) reduced reproduction and colonization in some species of brown algae (Duval et al. 1981). The results of laboratory studies are consistent with several of the effects observed following actual oil spills. Boney (1974) reported that aromatic hydrocarbons caused proliferations of tumor-like growths in marginal areas of *Porphyra* fronds, and increased apical growth of juvenile plants , although cell size was reduced. On the other hand, Steele (1977) reported that growth of Fucus zygotes was completely inhibited by 20 mg L^{-1} of No. 2 fuel oil, and growth and germination of zygotes ceased after exposures of reproductive fronds to only 0.2 ppb of No. 2 fuel oil or crude oil. The author concluded that Fucus spermatozoa were extremely sensitive to hydrocarbons during release and fertilization. Some field studies have also indicated that young plants may be sensitive to oil-contaminated substrates. For example, Thomas (1978) found that Fucus spiralis failed to survive and mature in oiled upper intertidal areas following the ARROW spill.

The magnitude of initial effects and the recovery rate of algal communities following oil spills is influenced by season, extent and duration of oil exposure, and species-specific sensitivities to petroleum hydrocarbons. Notini (1978) reported that algal diversity and abundance were not significantly affected following the IRINI spill and he suggested that this was due to the fact that most of the oil was removed before the beginning of annual spring growth, thereby reducing potential effects. Similarly, other investigators have suggested that effects of spills have been less severe when they occurred during periods of low algal growth (e.g. Ravanko 1972, cited in Notini 1978). Studies by Vandermeulen and Ross (1977), Hampson and Moul (1978) and Burns and Teal (1979) have indicated that marsh grasses exposed to oil retained in sediments have demonstrated slow recovery rates and reduced growth for several years. Burns and Teal (1979) found an inverse correlation between the amount of oil in surface sediments and plant growth, and also reported that areas with >1-2 mg oil g^{-1} sediment contained no living vascular plants. The following trends have also been identified by several investigators: (1) annuals are generally more sensitive than perennials (Baker and Crapp 1974, cited in Percy 1982; Hampson and Moul 1978); (2) fucoids may be less sensitive due to their mucilaginous coatings; and (3) some green and blue-green algae appear less sensitive than other intertidal algae than other species since they frequently are the first to colonize oiled substrates (Section 6.2.1.5).

6.2.1.4 Physiology and Metabolism

There have been few studies of the effects of petroleum hydrocarbons on the physiology and metabolism of intertidal algae. As indicated earlier, thin films of oil applied to the fronds of some algal species reduced gaseous diffusion rates and thereby caused a decrease in rates of photosynthesis (Johnson 1977). Baker (1970) suggested that oil exposure could lead to both accumulation of hydrocarbon degradation products and disruption of chloroplast membranes. Biosynthesis of nucleic acids in *Polysiphonia* and *Grateloupia* was inhibited by 36-h exposure to 1000 mg L^{-1} of crude oil (Davarin et al. 1975).

6.2.1.5 Habitat Loss and Disturbance

In most cases, the degree and duration of the loss of intertidal habitat for marine algae and marsh grasses is determined by the persistence of oil in substrates. In general, algal communities within low wave energy habitats in the upper intertidal may be affected by habitat loss and disturbance for longer periods than communities in high wave energy environments from which oil may be dissipated rapidly. Thomas (1977) reported that while *Fucus spiralis* settled on extensively oiled upper intertidal areas following the ARROW spill, the young plants did not survive. He also found reduced vertical distribution and biomass of some mid-intertidal species (*Chondrus crispus*, *Ascophyllum nodosum* and *Fucus vesiculosus*) in areas where oil persisted in sediments following this spill.

Oil remaining in marsh sediments may impede recolonization of marsh grasses that propagate through the development of new roots and rhizomes (Burns and Teal 1979). On the other hand, some genera of green algae (particularly *Ulva* and *Enteromorpha*) and blue-green algae appear to be more tolerant of oil-contaminated substrates, and may colonize oiled areas within two months after a spill (Duval et al. 1981). Hampson and Moul (1978) reported that a mat of blue-green algae developed following the BOUCHARD 65 spill, and suggested that increased light penetration due to lack of vegetation may have been responsible for its increased abundance.

6.2.1.6 Indirect Effects

Indirect effects of oil spills on plant communities result primarily from reduced grazing pressure or from changes in the species composition of the faunal component of the intertidal community. The most frequently observed indirect effect of past spills has been blooms of some green and blue-green algae (Duval et al. 1981). Such blooms have been documented following both fuel oil and crude oil spills (ELENI V, THUNTANK 6, BOUCHARD 65 and CHRYSSI P. GOULANDRIS accidents). Increases in the proportion of green to blue-green algae appear to be related partly to the greater tolerance of green algae to petroleum hydrocarbons.

6.2.1.7 Effects of Countermeasures

Countermeasures employed during the cleanup response to a number of oil spills may have contributed to the effects of oil on intertidal algae and marsh vegetation (Duval et al. 1981). Two of the major sources of impacts of cleanup responses on intertidal or marsh communities have been (1) cropping and incineration of oil-contaminated plants, and (2) direct application of chemical dispersants to intertidal habitats. These measures have been responsible in whole or in part for the mortality of intertidal or salt marsh flora. Various mechanical cleanup techniques (e.g. hydraulic cleaning) have also increased the local effects of some spills (Duval et al. 1981).

The direct application of chemicals to shorelines following the TORREY CANYON spill resulted in extensive mortality of intertidal algae (Southward and Southward 1978), which was attributed to the toxicity of the chemicals (largely detergents) used in intertidal habitats. Present guidelines for use of chemical dispersants in Canada do not recommend the use of these chemicals in shoreline areas (Environment Canada 1984). Chemical dispersants currently approved for use following oil spills are substantially less toxic to many marine fauna and flora than those employed following the TORREY CANYON accident and other oil spills from 1960 to 1972. Crothers (1983) reported no adverse effects on Fucus plots following an experimental application of weathered oil and dispersants. The effects of present generation dispersants and chemically-dispersed oil on other algal taxa are unknown.

The use of high-pressure hydraulic cleaning methods on oil-contaminated shorelines has also killed intertidal algae. Broman et al. (1983) reported greater mortality and slower recovery of intertidal algal communities in shoreline areas cleaned by high-pressure, hot water treatment than in adjacent oiled areas that were oiled but not cleaned. However, it should be noted that high-pressure hydraulic cleaning is generally used only in localized areas (e.g. piers, causeways, boardwalks) where the aesthetic impacts of oil contamination are considered undesirable.

Oil spills have had relatively major and long-term impacts on marsh habitats and vegetation, and cleanup operations have increased the effects of spills on these communities. The most serious impacts have resulted when heavy machinery has been used for removal of oil from contaminated marshes (Duval et al. 1981). Following the AMOCO CADIZ spill, mechanical removal of oil-contaminated sediments and marsh grasses resulted in major changes in drainage patterns in the Ile Grande marsh and prolonged the period required for recovery of vegetation communities (D'Ozouville et al. 1980). Vandermeulen and Ross (1977) investigated the effects of several counter-measures on a marsh affected by the GOLDEN ROBIN oil spill. Plots where oil and contaminated marsh vegetation were removed by hand spade and saw recovered more quickly than areas where vegetation was cut with sickles or scythes. Areas cleaned with mechanical devices such as sod-cutters required the longest recovery period, since these machines buried oil in the sediments, where it remained unweathered.

6.2.2 Vulnerability

The vulnerability of intertidal algae and marsh grasses to effects of oil spills varies widely depending on the circumstances surrounding the spill and the characteristics of the habitat and community affected. The most significant factors include: (1) type and volume of oil spilled; (2) the species of algae and its location in the intertidal zone; (3) extent of contamination of the intertidal zone; (4) season of year of the spill; (5) persistence of oil on the shoreline; (6) the physico-chemical nature of oil reaching the shoreline; and (7) the countermeasures utilized in the cleanup response. The relationships between most of these factors and the extent and duration of impacts of spills on intertidal and marsh vegetation have been discussed in preceding sections. Although some genera of algae appear to be more resistent to oil than others, algal communities in areas where oil persists for long periods of time are generally most vulnerable to effects of oil spills.

SECTION 6.3

INTERTIDAL FAUNA

Linda Martin

There is a considerable database regarding the effects of oil on intertidal invertebrates since most oil spills have occurred in nearshore environments and have contaminated intertidal areas to some degree. This information includes documented *in situ* effects of actual spills on intertidal communities, laboratory studies of the toxicity of various hydrocarbons to different species and, more recently, investigations of the sublethal effects of oil metabolic, physiological or behavioural activities and recovery time. Many of these studies have been previously reviewed by Percy and Mullin 1975; Malins 1977; Percy 1982; Duval et al. 1981; and Engelhardt 1985.

Intertidal faunal assemblages vary considerably between temperate and cold water marine areas. In the arctic, a few mobile species from subtidal habitats may temporarily occupy the intertidal area. However, in more temperate regions a wide range of groups numbering many hundreds of species may be present in the intertidal zone. Moreover, the number of species may vary considerably between areas and from season to season. Typically fewer groups occur in winter because some species migrate to subtidal areas at this time. All these variables, as well as a number of physical factors, influence the effects of oil exposure in this complex habitat. The effects of oil spills on intertidal fauna are often difficult to distinquish from the effects of these events on the entire community comprising both fauna and flora together. The effects of petroleum hydrocarbons on intertidal algae and marsh grasses discussed in the previous section will influence both the initial impact and recovery of fauna in this habitat.

In the following section, the effects of oil on gastropods and sessile crustaceans (e.g. barnacles) are discussed. Numerous other taxonomic groups (e.g. bryozoans, nemerteans, hydroids) occupy this diverse habitat, but are generally less conspicuous within the community. Little information on the effects of oil on these species is available in the literature. Readers are referred to Sections 4.3 and 4.4 for information on the effects of oil on benthic bivalves, crustaceans and polychaetes in subtidal habitats. Many of these species also occupy the intertidal zone and the effects of oil on these groups would likely be similar, although physical smothering is more likely to occur in the intertidal zone.

6.3.1 Sensitivity to Oil

6.3.1.1 Mortality

Following a spill, the most common cause of mortality among many intertidal species is smothering by thick layers of stranded oil, which interfers with feeding, respiration, and attachment to the substrate, and causes severe habitat disturbance. Often there is complete elimination of intertidal fauna (particularly periwinkles, limpets, tube worms, barnacles, and crustaceans) in heavily oiled areas (Duval et al. 1981). Smothering is a common cause of death in sessile organisms such as barnacles and tube worms, which cannot shed or avoid thick layers of stranded oil. Other organisms such as mussels (*Mytilus* sp.) are often more resistant to suffocation by oil, and in some cases, have been reported to suffer less mortality than barnacles. The loss of limpets and snails in heavily contaminated areas has been attributed to suffocation, although these animals have also been observed alive despite extensive oiling of their shells. In addition, mortality of periwinkles and some limpets has been caused by detachment from the substrate. In the field, toxic effects of hydrocarbons are often indistinguishable from the physical effects of oil, although direct toxicity of oil to barnacles (Foster et al. 1971), sea anemones (Clark and Finley 1977), and mussels (Straughan 1976) has been documented.

Numerous laboratory investigations of the toxic effects of petroleum hydrocarbons on invertebrates have been reviewed by Craddock (1977); Percy (1982); and Wells and Percy (1985). The results of bioassays conducted with crude and bunker oils have suggested that mortality due to toxicity is relatively high, but few groups other than molluscs and crustaceans have been studied. Toxic effects have been documented in adult invertebrates as well as the larval and juvenile stages of many species. Unfortunately, laboratory bioassays generally suffer from a lack of standardized methods, and this complicates interpretation and extrapolation of results to actual spill circumstances. Most investigations of oil toxicity in intertidal organisms have been restricted to studies with species found in lower intertidal or shallow subtidal habitats. Although intertidal fauna, particularly those species occupying habitats in the upper intertidal zone, are more susceptible to physical effects such as suffocation, they tend to be more tolerant to toxic effects of oil than subtidal or pelagic fauna (Rice et al. 1979, Table 6.3-1). Their lower sensitivity to the acute toxic effects of oil may be explained by the fact that upper intertidal fauna are generally well adapted to natural conditions of short-term physical stress (e.g. resistance to desication). Other generalizations based on laboratory toxicity data include the greater sensitivity of arctic and subarctic species and larvae and juvenile stages to oil than temperate species and adults, respectively (Rice et al. 1977a; Percy 1982).

The persistence of oil in cold water environments, has been cited as the probable cause of the greater sensitivity of arctic and subarctic invertebrates, rather than any intrinsic species-specific differences in sensitivity (Craddock 1977; Percy 1982). It is also generally agreed that soluble aromatic hydrocarbons, particularly the naphthalenes, may be the most acutely toxic constituents of crude oils and refined petroleum products (Craddock 1977). Because some refined fuels such as diesel oil contain relatively greater proportions of aromatic hydrocarbons than crude oils, they are generally considered more acutely toxic. The reasons for the greater sensitivity of larval and juvenile stages to oil are not well understood, but may be due to less developed hydrocarbon depuration processes or the need to direct most energy expenditure to growth rather than depuration or detoxification of petroleum hydrocarbons.

TABLE 6.3-1

RANGES OF SENSITIVITIES FOR SIMILAR GROUPS FROM HABITATS EXPOSED TO COOK INLET CRUDE OIL

Organism group	96-h LC_{50} (mg L^{-1} WSF) Habitat		
	Pelagic	Benthic	Intertidal
Crabs and Shrimps	1 - 5	3 - 5	8 - >10
Molluscs	-	4 - >8	8

Adapted from Rice et al. 1979

Although intertidal invertebrates have been reported to be the most tolerant group in short-term bioassays with crude oil WSFs (Rice et al. 1979), more recent studies of the effects of long-term exposure to aromatic hydrocarbons at concentrations below the 96-h and 24-h LC_{50} levels have shown delayed mortality in some species. Stickle et al. (1984) reported that the LC_{50} for Thais lima and Mytilus edulis with the WSF of Cook Inlet crude oil declined with duration of exposure. For example, the LC_{50} for T. lima was initially greater than 3000 ppb (aromatic hydrocarbons), but declined to 818 ± 118 ppb after 28 days of exposure. In the same period, the LC_{50} for M. edulis declined from >3000 ppb to 1689 ± 42 ppb. Based on field investigations of the METULA spill, Straughan (1976) reported delayed mortality of Mytilus sp. due to toxicity of crude oil. Stickle and his colleagues have attempted to determine the causes of delayed mortality, and their conclusions are discussed in Section 6.3.1.3.

6.3.1.2 Growth, Reproduction and Development

Petroleum hydrocarbons have been shown to have adverse effects on weight per unit length ratios, byssal thread production, and reproduction. Thomas (1978) found that six years after the ARROW Bunker C spill, periwinkles (*Littorina littorea*) in areas affected by the spill showed significantly lower increases in weight per unit of length than those at control stations, and this was accompanied by higher hydrocarbon concentrations in body tissues. On the other hand, the body weights of mussels (*Mytilus edulis* and *M. californianus*) in areas affected by the Santa Barbara blowout were apparently unaffected in the three years following the event (Harger and Straughan 1972). Byssal thread production in mussels is very sensitive to oil exposure and Rice et al. (1983) found a good correlation between sublethal oil concentrations and the number of byssal threads produced.

Adverse effects of oil on the reproduction of most groups of intertidal fauna have been observed in laboratory and field investigations. Field studies of a fuel oil spill in the Wild Harbour River estuary, Massachusetts, indicated that immediately after the spill, surviving fiddler crabs (*Uca pugnax*) showed breeding colouration out of season and moulted (Sanders et al. 1980). Recruitment to this crab population was unsuccessful for eight years after the incident, and Krebs and Burns (1978) showed that recovery of the population was highly correlated with the disappearance of the naphthalene fraction of aromatics in marsh sediments. Although adults produced young in the years after the spill, mortality of juveniles resulted in a loss of recruitment to the population. Sanders et al. (1980) reported that juvenile crabs were killed at oil concentrations of 100-200 mg L^{-1} in the sediments, and that sublethal effects occurred at lower concentrations. The mating behaviour of other species of shore crabs (*Pachygrapsus crassipes*) was disrupted in the presence of only 10 ppb of the soluble fractions of crude oil (Takahashi and Kittredge 1973, cited in Johnson 1977). The reproduction and survival of the young of some gastropods has also been adversely affected by oil. For example, Kuiper et al. (1984) reported inhibition of reproduction in Littorina littorea and reduced survival of larval Hydrobia ulvai following exposure to petroleum hydrocarbons.

Petroleum hydrocarbons also affect reproduction of sessile intertidal fauna. In barnacles, this may vary among species. No effects have been reported in the high intertidal barnacles Balanus glandula and Chthamalus fissus, but reduced reproduction has been observed in goose neck barnacles (Pollicipes polymerus) found in the lower intertidal (Straughan 1970). Not only was reproduction of adult Pollicipes polymerus affected, but larvae of this species (which set preferentially on the stalks of adults) would not set on oil-contaminated adults (Straughan 1970). Straughan (1970) also found that oil inhibited settling of B. glandula and C. fissus larvae, but B. glandula appeared to be slightly less affected (faster recolonization rate) than C. fissus. Both inhibition and reduction of reproduction has been reported in mussels (Mytilus edulis and M. californianus) exposed to oil in intertidal habitats (Straughan 1970; Sanders 1977). The results of an experimental 140-d exposure of mussels to low concentrations of hydrocarbons (30 μg L^{-1}) indicated no adverse effects on the production or maturation of ripe gametes (Widdows et al. 1982). However, Gilfillan (1975) found that mussels exposed to low concentrations of petroleum hydrocarbons had lower feeding and assimilation rates and elevated respiration rates and this resulted in a net reduction in carbon flux that could adversely affect reproduction and growth.

6.3.1.3 Metabolism and Physiology

Adverse effects of hydrocarbons on various physiological processes in intertidal invertebrates have been documented, but the majority of the studies have focused on mussels. Some of the important observed effects include: changes in oxygen consumption rate; reduced food absorption efficiency; and changes in the enzyme activity associated with digestive cells and the gills.

These effects are discussed in more detail below. In some physiological processes, adverse effects of hydrocarbon exposure have not been noted. For example, enzyme activity in the mantle of mussels and enzyme activities in oysters were reported to be unchanged in long-term exposure studies conducted by Widdows et al. (1982) and 7-d exposure studies of Heinz et al. (1974, cited in Varanasi and Malins 1977).

Stickle et al. (1984) investigated the Scope for Growth, predation rate and energy expediture (respiration, ammonia excretion and primary amine loss) of the carnivorous snail Thais lima and its prey Mytilus edulis, during a 28-d exposure to the WSF of Cook Inlet crude oil. These researchers found that the Scope for Growth varied inversely with the concentration of WSF and became negative at concentrations greater than 19 to 23 percent of the 28-d LC_{50}, which indicated that the snails would eventually die. Stickle et al. (in press, cited in Stickle et al. 1984) also noted a negative relationship between Scope for Growth and aromatic hydrocarbon concentration in M. edulis. Variations in the feeding rate of T. lima followed the same trend as Scope for Growth, and were the primary reason for the difference in the Scope for Growth. The total energy expediture rate of T. lima also declined with increased WSF concentration, although variability in oxygen consumption rates rather than maintenance costs associated with excretion accounted for most of the observed differences in energy expenditure (Stickle et al. 1984).

Oxygen consumption appears to be a very sensitive indicator of stress in molluscs, but the direction of changes in respiration vary with the hydrocarbon concentration. The rate of oxygen consumption by Thais lima increased during 28-d exposures to 44 ppb of a WSF (approximately equal to 5 percent of the 28-d LC_{50}). Low oil concentrations have also caused a similar increase in respiration of mussels (Gilfillan 1975; Widdows et al. 1982). However, lethal concentrations of the WSF of crude oil depressed the oxygen consumption rate of T. lima (Stickle et al. 1984) and some bivalves (Gilfillan 1975; Widdows et al. 1982).

The research of Gilfillan (1975) indicated that growth of mussels exposed to oil could be reduced due to a decrease in net carbon flux resulting from an increase in respiration rates and reductions in assimilation and feeding rates. Reduced cirral activity of barnacles and depressed ciliary activity of mussels have also been reported during exposure of these organisms to crude oil dispersions. Some authors have also suggested that reduced feeding rates would adversely affect growth of organisms if exposure to hydrocarbons was prolonged (Smith 1970; Dunning and Major 1974).

Widdows et al. (1982) reported that mussels exposed to water-soluble hydrocarbons reduced their clearance rates. Biochemical changes noted by Widdows et al. (1982) included increased carbon flux through the glycolytic pathway and destabilization of lysosomes in digestive cells.

6.3.1.4 Uptake and Biotransformation

The uptake and concentration of petroleum hydrocarbons in the tissues of the mussel Mytilus edulis has been documented in numerous field and laboratory studies (e.g. Ganning et al. 1983; Boehm et al. 1982; Widdows et al. 1982; Straughan 1981; Grahl-Nielsen et al. 1978; Vandermeulen and Penrose 1978). This bivalve is utilized throughout the world as a sentinel organism to monitor levels of contaminants in estuarine and coastal environments. Hydrocarbon accumulation in mussel tissues to concentrations well above those in the surronding water has been documented during several studies (e.g. of hydrocarbons in mussel tissues) (Fossato and Cansonier 1976, cited in Ganning et al. 1983; Widdows et al. 1982). In some instances, concentrations have been more than 1000 times greater than the exposure level. Uptake is initially rapid, and is followed by a period of gradual accumulation during long-term exposures (Widdows et al. 1982). High-resolution gas chromatograms of mussel tissues following the TSESIS spill indicated that the hydrocarbons initially accumulated by these organisms were similar in type to those present in oil from to the TSESIS. However, relatively rapid changes in the oil composition within tissues occurred. N-alkanes were preferentially degraded over the corresponding branched and isoprenoid compounds (Boehm et al. 1982). On the other hand, Stickle et al. (1984) reported that naphthalene and its derivatives accounted for most of the measured aromatic hydrocarbons that were accumulated in the soft tissues of *M. edulis* and *Thais lima* during 28-d exposures to WSFs of crude oil. Other investigators have also documented differential uptake of hydrocarbons from water by molluscs. For example, Neff et al. (1976) reported the following differential hydrocarbon uptake sequence: naphthalene < methylnaphthalenes < dimethylnaphthalenes.

Grahl-Nielsen et al. (1978) examined the uptake and depuration of oil by intertidal fauna over a one-year period following a crude oil spill in Norway. Crabs contained less aromatics than other species, and limpets accumulated smaller quantities of aromatic hydrocarbons than periwinkles. This study also demonstrated preferential retention of dibenzothiophenes in mussels and periwinkles and the concomitant loss of naphthalenes from tissues during the year after the spill. Boehm et al. (1982) reported a similar pattern of retention of phenanthrene and dibenzothiophene and depuration of naphthalene in mussels during the year following the TSESIS spill. Widdows et al. (1982) and Clement et al. (1980) noted that the accumulation of hydrocarbons appeared to be controlled by decreasing water-solubility with increasing alkyl substitution, rather than by the abundance of specific hydrocarbons in the water-soluble fraction.

Widdows et al. (1982) examined the relative concentrations of hydrocarbons in the tissues of mussels (*Mytilus* sp.) exposed to 30 $\mu g\ L^{-1}$ oil for five months. Levels of aromatic hydrocarbons in the digestive gland were approximately five times higher than those measured in other tissues, and it was concluded that this trend was at least partly due to the higher lipid concentration in the digestive gland. Other researchers have also identified the lipid content of tissues as an important factor that determines the potential for hydrocarbon accumulation (Stegenian and Teal 1973, Fossato and Canzonier 1976, both cited in Widdows et al. 1982; Varanasi and Malins 1977). Widdows et al. (1982) also found that digestive gland tissues accumulated aliphatic hydrocarbons more rapidly than aromatic fractions. However, aromatic fractions gradually increased and the concentration of aliphatic hydrocarbons decreased throughout the experimental exposure period.

Depuration of hydrocarbons in mussels and other molluscs has been reported when individuals experimentally or accidentally exposed to oil are placed in clean sea water (Ganning et al. 1983; Notini 1980, Fossato and Canzonier 1976, both cited in Ganning et al. 1983; Straughan 1981; Grahl-Nielsen et al. 1978; Varanasi and Malins 1977). Until recently, little was known regarding the mechanism of hydrocarbon depuration, and several investigators had reported that increased mixed-function oxygenase (MFO) or aryl hydrocarbon hydroxylase (AHH) activity did not occur in mussels (Vandermeulen and Penrose 1978; Lee et al. 1972, Payne 1976, both cited in Vandermeulen and Penrose 1978; Varanasi and Malins 1977). However, increased activity of these enzyme systems is now believed to account for the formation of virtually all of the primary metabolic products of aromatic hydrocarbon degradation (Varanasi and Malins 1977). Recently, Moore et al. (1980) confirmed that *M. edulis* could synthesize specific enzyme components of the mixed function oxygenase system in response to hydrocarbon exposure, and Widdows et al. (1982) indicated that *M. edulis* was "capable of metabolizing hydrocarbons, albeit at a low rate".

Due to the relatively slow rates of hydrocarbon depuration by molluscs, several authors have suggested the possibility of food chain biomagnification. However, the results of one study indicated that this may not be a justifiable area of concern. Stickle et al. (1984) examined hydrocarbon concentrations in tissues of *Thais lima* that had been fed *Mytilus edulis* exposed to a 583 ppb WSF for 28 d, and did not find a significant relationship in tissue concentrations between the two organisms that would provide evidence of biomagnification.

6.3.1.5 Pathology and Histology

There have been very few investigations of the pathological or histological effects of oil in intertidal fauna. Widdows et al. (1982) found smaller epithelial cells in the digestive tubules of mussels exposed to 30 μg L^{-1} of North Sea crude oil for 140 d, and a concomitant change in the shape of the lumens from convoluted to round. They also noted an increase in the number and size of vacuoles within the digestive cells. Although Straughan (1970) had reported earlier that reproductive tissues of Mytilus were flaccid and contained few eggs following the Santa Barbara blowout, Widdows et al.(1982) did not document adverse effects of hydrocarbon exposure on development of gametic tissues of mussels.

6.3.1.6 Behaviour

The effects of oil on the behaviour of molluscs (gastropods and bivalves) and crustaceans were reviewed by Johnson (1977). One of the major effects noted was narcotization. Other effects included: (1) loss of pedal attachment (limpet, Patella vulgata; Dicks 1973); (2) increased crawling rate (snail, Littorina littorea; Hargrave and Newcombe 1973); (3) reduced activity level (snail, L. saxatilis; Baker 1973); and (4) reduced chemotaxis to prey (snail, Nassarius obsoletus; Jacobson and Boylan 1973, oyster drill, Urosalpinx cinerea; Eisler 1975). Reproductive and feeding behaviour of many arthropods are also dependent on chemosensory inputs. The results of numerous studies summarized by Johnson (1977) indicate that there has been a wide range of behavioural reactions to oil depending on the test species, exposure concentrations and durations, and hydrocarbon fractions examined. Kittredge et al. (1974, cited in Johnson 1977) described the results of studies with crabs and suggested that exposure to oil destroyed the dendritic tips in the aesthetascs (sensory structures which are thought to facilitate chemosensory reception). Studies of the fiddler crab (Uca pugnax) population in the Wild Harbour River estuary also indicated that several behavioural effects were attributable to the FLORIDA fuel oil spill (Sanders et al. 1980). These behavioural effects, combined with other lethal and sublethal effects, reduced the crab population for at least seven years after the incident (Sanders et al. 1980). Sanders et al. (1980) summarized the results of a number of investigations on behavioural effects in Uca pugnax as a result of oil exposure. These included: (1) loss of equilibrium; (2) increased moulting, (3) display of breeding colouration and activities during the non-reproductive season; (4) abnormally shallow burrowing depths that allowed winter freezing; and (5) lethargic activity which resulted in impaired escape responses.

6.3.1.7 Habitat Loss and Disturbance

Long-term studies conducted following the METULA, FLORIDA and ARROW oil spills all indicated that the persistence of oil in intertidal habitats determined the time required for recovery of fauna. Analysis of data collected one year and three years after the METULA spill indicated that the presence of crude oil in sediments was the factor with the most significant and consistent relationship to the distribution and abundance of marine invertebrates (Straughan 1981). Sanders et al. (1980) were also able to correlate both the magnitude and duration of effects of the FLORIDA spill on intertidal invertebrates with the relative degree of oil contamination at various sites in Wild Harbour River estuary. Thomas (1978) reported similar relationships between oil persistence and effects on invertebrates in Chedabucto Bay after the ARROW spill. Fauna in intertidal areas receiving little oil have consistently been shown to be the least affected and recover most quickly following spills, whereas areas that are extensively contaminated with oil are typically much slower to recover in terms of the distribution and abundance of fauna. Since recovery depends on the removal of oil from shoreline areas through physical processes (e.g. flushing by wave and tidal action) and biodegradation, the duration of habitat disturbance or loss depends on the rate of oil removal. This process depends on a number of factors including the degree of exposure to waves, type of substrate, volume and type of oil reaching the shoreline, physico-chemical characteristics of oil following weathering (both at sea and on the shore), time of year, degree and extent of intertidal contamination and the shoreline restoration methods employed during any cleanup response. The relative persistence of oil in various types of intertidal habitats and hence the relative rates of recovery are indicated in Table 6.3-2, which was adapted from a study by D'Ozouville et al. (1981) following the AMOCO CADIZ spill. It is important to emphasize that this table depicts a very general relationship among habitats based on the results of a number of processes, and depending on the circumstances surrounding a particular event, the persistence of oil in many different intertidal habitats could be ten years or longer.

6.3.1.8 Indirect Effects

Indirect effects of oil spills on the habitat of intertidal fauna may occur due to the sensitivity of some intertidal algae to hydrocarbon exposure (Section 6.2) and the potential for substantial changes to the physical-chemical nature of intertidal substrates following oil contamination. Intertidal algae provide a source of food and cover for numerous intertidal species, and when algae are damaged by oil or removed from the shore during the cleanup response, the organisms that rely on these plant communities will be affected. Grazers such as limpets that remain in a contaminated area but are unaffected by the oil itself, may still be indirectly affected because of reduced food availability. Smaller crustaceans, gastropods and other taxa that rely on algae for cover and protection from wave action and predators can also be indirectly affected by oil spills. On the other hand, algal blooms may follow oil spills when extensive mortality of intertidal fauna occurs, and this growth may eventually enhance the recolonization of the area by grazers, if populations of these invertebrates are in adjacent areas unaffected by the spill.

Depending on the type, volume and physico-chemical characteristics of oil reaching the shoreline and the cleanup techniques employed, changes in the character of the intertidal substrate can result. For example, following the METULA spill, water-in-oil emulsions (Mousse) hardened in intertidal substrates and formed an asphalt-like substance that was slowly eroded away by wave action (Straughan 1981). This erosion was a slow process, although recolonization of the newly exposed shoreline by intertidal organisms proceeded

TABLE 6.3-2

PERSISTENCE OF OIL IN INTERTIDAL SUBSTRATES*

AND APPROXIMATE TIME SCALE FOR REMOVAL

DECREASING OIL PERSISTENCE	
≥ 10 years	SHELTERED MARSHES AND LAGOONS
	SHELTERED TIDAL FLATS
	SHELTERED ROCKY AREAS AND POCKET BEACHES
	GRAVEL REACHES
	MIXED SAND AND GRAVEL BEACHES
	EXPOSED TIDAL FLATS
	COARSE SAND BEACHES
	EXPOSED FINE-SAND BEACHES
	ERODING WAVE-CUT PLATFORMS
≤ 1 year	EXPOSED ROCKY HEADLANDS

*AFTER D'OZOUVILLE et al. 1981

with the removal of the stranded oil masses. Habitat changes from sand to "asphalt" had a major influence on the species that were capable of recolonizing the area, and as a result, recovery to pre-spill conditions was delayed until most of the stranded oil was removed. Oil may also cause the agglomeration of loose cobble and gravel, thereby preventing cryptic fauna from inhabiting affected areas. Other indirect effects may result from certain types of shoreline cleanup activities and are discussed in the following section.

6.3.1.9 Effects of Countermeasures

Countermeasures employed during the cleanup of a number of past oil spills have contributed to the effects of the event on intertidal fauna (Duval et al. 1981). Cleaning of shores with toxic first-generation dispersants (nonionic surfactants with a high aromatic content solvent stabilizer; Southward and Southward 1978), heavy machinery that disturbed the habitat and removed algae and fauna (D'Ozouville et al. 1980), and high-pressure hot water cleaning (Broman et al. 1983) all had adverse effects on shoreline fauna and may have been responsible in part for the relatively long recovery periods. Recent field investigations of various shoreline restoration methods and summaries of techniques applicable to various types of shorelines (Sergy, in press; CONCAWE 1983; Vandermeulen and Ross 1977; Owens 1977) have greatly improved available information regarding the effects of countermeasures on different intertidal habitats. The dispersants now available are less toxic than first-generation dispersants to marine fauna, and Environment Canada (1984) guidelines restrict their use on shorelines. When present knowledge regarding the effects of countermeasures is applied to oil spill cleanup responses, some of the adverse impacts observed in the past should be avoidable.

6.3.2 Vulnerability

The vulnerability of intertidal fauna to oil spills is influenced by all of the factors related to oil characteristics and the intertidal environment that were discussed earlier in Section 6.1 and 6.2.2. Sessile species such as barnacles and mussels are most vulnerable to suffocation, but may be less sensitive to the toxic effects of oil than other species and, therefore, may be more capable of survival following a spill. Mobile species such as limpets and snails, although they are vulnerable, may be able to tolerate moderate oil exposures if adjacent uncontaminated habitat is available. In general, intertidal habitats are exceedingly vulnerable to oil exposure, but their recovery potential varies considerably after contamination. The most sensitive habitats, and consequently species that occupy these habitats, are those in fine-to coarse-grained substrates in low wave energy areas where oil can persist for prolonged periods. On the other hand, exposed rocky intertidal habitats tend to recover quickly and may therefore be considered less vulnerable to effects of oil spills.

SECTION 6.4

MARINE-ASSOCIATED MAMMALS

Wayne Duval

The effects of oil on truly marine mammals, including whales, seals, porpoises and walruses were described in earlier sections of this report (Section 2.3 and 2.4). The following sections summarize available information describing the fate and effects of oil on mammals associated with shoreline or ice-edge habitats, but frequently found in marine or estuarine waters. The majority of the literature was recently summarized by Duval et al. (1981); ESL (1982a) and Richardson et al. (1983). It includes experimental studies of sea otters (*Enhydra lutris*) and polar bears (*Ursus maritimus*), as well as documentation of effects on sea otters, lemmings and European otters (*Lutra lutra*). Although the *L. lutra* is not found in Canada, information on this mammal is included here because the river otter (*Lutra canadensis*) in Canada occupies similar habitats, including both estuaries and marine waters.

6.4.1 Sensitivity to Oil

6.4.1.1 Mortality

Mortality of marine-associated mammals as a result of oil spills may be relatively rare. In some cases, deaths have not been unequivocally linked to oil exposure. Mortality has been at least in part attributable to the lack of actual contact with oil, despite the fact that various laboratory and field experiments have shown that some mammals associated with shoreline habitats can be seriously affected by oil contamination.

Ramseier et al. (1973) observed a single dead lemming following a diesel and gasoline spill from a storage tank farm at Deception Bay in northern Quebec, and suggested that other lemmings may have been killed as a result of this event. However, these authors also stated that the avalanche that caused the rupture of storage tanks, rather than exposure to the fuel, may have been responsible for the single confirmed death. A Bunker C spill from the ESSOBERNICIA in the Shetland Islands resulted in the death of 20 to 30 otters, and indirectly contributed to the death of 50 domestic sheep that ingested oiled seaweed when snowfall at higher elevations forced the herd to feed in shoreline areas (Bourne 1979). Baker et al. (1981, cited in Richardson et al. 1983) reported mortality of European otters following exposure to bunker fuel and ingestion of oil-contaminated prey, while Barabash-Nikiforov et al. (1947, cited in Richardson et al. 1983) reported that a gasoline and diesel oil spill in the Duril Islands near the coast of the Soviet Union killed sea otters.

Mortality of sea otters and polar bears has also been documented in field experiments. For example, one of the sea otters died during studies of the effects of oil contamination on metabolic rates (Costa and Kooyman 1981). Two of three captive polar bears also died 26 and 27 d after immersion in oil-covered water for 15-20 min (Engelhart 1981; Oritsland et al. 1982). Each of these investigations is discussed below in further detail.

6.4.1.2 Behavioural Effects

A range of behavioural effects have been observed following experimental or accidental exposure of otters and polar bears to oil, although these responses have often varied between animals and with the circumstances surrounding the oil exposure. Avoidance of oil by sea otters is a variable behavioural response. For example, Barabash-Nikiforov et al. (1947, cited in Richardson et al. 1983) reported that sea otters avoided shorelines contaminated with gasoline and kerosene, while Baker et al. (1981, cited in Richardson et al. 1983) observed one European otter walking through oil-contaminated shoreline areas and another diving and emerging in an area containing Bunker C on the water surface. Captive sea otters have also been shown to enter an area covered with Prudhoe Bay crude oil when adjacent areas were uncontaminated (Siniff et al. 1982). During the latter experiment, otters repeatedly entered the contaminated area and surfaced through the oil, but immediately dove when the surface slick was contacted.

Other behavioural responses observed in captive or wild sea otters have included: (1) grooming of oiled fur both underwater (Williams 1978, cited in Richardson et al. 1983) and on the surface (Baker et al. 1981); (2) continuous shivering (Kooyman and Costa 1978); (3) increased activity of animals for one week following application to fur of a small quantity (25 mL) of Prudhoe Bay crude oil (Siniff et al. 1982); and (4) possible ingestion of oil-contaminated seabirds, and subsequent death (Baker et al. 1981).

Exposure of free-ranging polar bears to oil in shoreline areas, ice-edge habitats or adjacent marine waters has not been observed to date. However, the previously mentioned experiments conducted with three captive polar bears provide some information on the behavioural responses of this species to oil. Three bears were forced into an oil-covered pool (Midale crude), and after leaving the pool, actively ingested oil by licking it off of their fur and occasionally from the cage walls (Oritsland et al. 1982). During the recent public hearings related to Beaufort Sea hydrocarbon development, one participant stated that there have been instances where polar bears have entered northern camps, broken into fuel storage areas, and subsequently ingested motor oil after crushing the cans in their jaws (R. Hurst, INAC, pers. comm.). These observations clearly suggest that this species is unlikely to avoid oil or oil-contaminated prey.

During these experiments, grooming activity increased after the fur was contaminated with oil, and the presence of oil in faeces for the next 4 weeks suggested that considerable quantities of oil were ingested during this time (Engelhardt 1981). The bears also vomited oil during the day after oil exposure. Other behavioural reactions noted by the research team were tremors and anorexia (Engelhardt 1981).

6.4.1.3 Physical Effects

The physical effects of oil exposure on the thermal conductance of otter and polar bear pelts have been investigated in laboratory experiments.

Costa and Kooyman (1981) reported that oil contamination increased the wetability of sea otter fur, and showed that this effect could persist even after the oil had been removed. The thermal conductance of pelts from two adult sea otters increased by factors of 1.1 and 1.3 following application of Prudhoe Bay crude oil, and thermal conductance of a pelt from a pup increased by a factor of 2.1 (Kooyman et al. 1977). As indicated in Section 6.5.5, increased thermal conductance of a sea otter pelt decreases its insulative capacity, and this may lead to chilling and subsequent death.

The effects of oil contamination on the thermal conductance of polar bear pelts have also been examined experimentally with captive bears. As in sea otters, the fur is important for thermal insulation of polar bears, although it is supplemented by subcutaneous fat and a stagnant layer of water adjacent to the skin when the fur is wetted (Fritsch et al. 1974, cited in Richardson et al. 1983). Hurst and Oritsland (1982) found that the thermal conductance of polar bear fur in air was increased after oiling with Midale crude oil, 10W motor oil and 'Base 2500' oil. Exposure of the captive bears to Midale crude also resulted in marked epidermal responses and acute inflammation of the nasal passages, probably due to the local irritant action of crude oil (Oritsland et al. 1982). Thermoregulation studies were also conducted with the three live bears that were immersed in pools containing Midale crude oil (Section 6.4.1.2).

A polar bear with an oily substance on its neck and shoulder was recently observed near Churchill, Manitoba (I. Stirling, pers. comm.). One apparent effect of this contamination was the loss of fur near the unidentified oily substance. Samples of oiled fur were taken from this animal to identify the contaminant and to determine the possible reason for fur loss, although the results of these analyses are not yet known.

6.4.1.4 Effects on Growth, Development and Reproduction

The effects of oil exposure on growth, development and reproduction of marine-associated mammals have not been directly examined. However, many of the physiological effects described in the subsequent section would have implications for growth and perhaps reproduction in otters and polar bears.

6.4.1.5 Physiological Effects

A substantial amount of information on the physiological effects of oil on otters and polar bears has resulted from a relatively limited number of investigations. Sea otter metabolic rates increased after oiling by factors ranging from 122 to 169 percent (Costa and Kooyman 1981). During these studies, only a small quantity of oil (35-60 mL) was applied to 25 percent or less of the fur on each test animal, suggesting that increases in metabolic rate would be even more pronounced if the degree of contamination were more extensive. The increased metabolic rates were accompanied by a reduction in subcutaneous temperatures by about 10 °C relative to control animals. The authors suggested that these reductions in temperature were a result of peripheral vasoconstriction in an attempt to minimize heat loss through the oiled fur. Costa and Kooyman (1981) concluded that oil contamination resulted in a profound increase in the heat loss of sea otters, and expected that contamination of 30 percent or more of the fur would probably result in mortality.

There may be other physiological effects of oil on sea otters, but these are largely undocumented. Williams (1978, cited in Richardson et al. 1983) reported that a single otter that died shortly after removal from oily water had congested tissues and mild focal enteritis, although this was not confirmed to be the cause of death or to be related to previous exposure to oil. Autopsies conducted on five oiled European otters indicated that death of all of the animals was due to hemorrhagic gastro-enteropathy; four of the otters also had oil present in their intestines (Baker et al. 1981).

As indicated earlier, studies of thermoregulation in three captive polar bears following oil contamination were conducted in conjunction with other investigations (Hurst et al. 1982; Oritsland et al. 1982).

Physiological effects observed included increased metabolic rates during both rest and exercise, and elevated skin temperatures. The latter effect was opposite to the response documented in sea otters by Costa and Kooyman (1981). The authors explained this phenomenon as a skin reaction (presumed to be vasodilation) that was overriding the normal thermoregulatory response (Hurst et al. 1982; Oritsland et al. 1982).

Other physiological effects of oil on polar bears documented during this investigation were: (1) neurological disorders presumably associated with the uptake of low molecular weight hydrocarbons from the ingested oil; (2) anemia resulting from hemolysis and lack of counterbalancing blood-forming responses by bone marrow; (3) major decreases in red blood cell count, hematocrit and hemoglobin concentration; (4) increases in serum bilirubin levels because of the hemolysis; (5) changes in serum electrolyte levels and plasma hormone levels due to stress; and (6) high levels of urea nitrogen in the serum as a result of renal impairment (Hurst et al. 1982; Oritsland et al. 1982).

6.4.1.6 Pathological Effects

Pathological effects of oil exposure, such as induction of tumorous growths and neoplasms, increased frequencies of disease and parasitic infections. Other abnormalities documented in fish and invertebrates have not been observed in otters and polar bears exposed to oil. However, this may have been a function of the specific interests of the investigators, rather than an absence of such pathological effects.

6.4.1.7 Hydrocarbon Uptake, Metabolism and Depuration

The fate of oil ingested by otters has not been examined, although some hydrocarbon uptake through the gut probably occurs because oil has been observed in the intestinal tract (Baker et al. 1981). At present, there is no evidence to suggest that hydrocarbon uptake and depuration pathways or sites of accumulation in the tissues of otters would be different than those documented in seals (Section 2.3) and polar bears.

Engelhardt (1981) provided some information on the fate of oil ingested by two polar bears. The proportions of low molecular weight hydrocarbons measured in vomitus and faeces were lower than in the oil ingested by the bears, suggesting that these fractions were most readily absorbed. Although oil levels in plasma decreased substantially 1 to 2 weeks after the bears were contaminated, hydrocarbon concentrations in bile and urine remained comparatively high up to 4 to 5 weeks after oiling. Engelhardt (1981) suggested that this trend was indicative of depuration by these pathways (e.g. the excretory system). Hydrocarbon concentrations were also measured in various tissues of the two bears that died after 4 weeks, and were highest in kidney, brain and bone marrow, but the relative amounts of hydrocarbons in each of these tissues were not the same in both bears.

6.4.1.8 Effects of Oil Spill Countermeasures

There is no available information on the direct effects of oil spill countermeasures on marine-associated mammals, or on the indirect effects of various cleanup techniques in reducing or increasing the impacts of oil. However, given the range of effects that can result from contamination of the fur of these species, it is likely that any countermeasure that reduced the potential for oil exposure in either water surface or shoreline habitats would decrease the potential for serious effects of oil spills on marine-associated mammals.

Although not strictly an oil spill countermeasure, there is some limited information on the effectiveness of efforts to remove oil from sea otters. Kooyman and Costa (1978) reported that one sea otter that was experimentally oiled and then cleaned was unable to groom its fur to a normal condition, but shivered constantly at a relatively warm water temperature (16.7 °C). In a later study, the same researchers noted that metabolic rates of an oil-contaminated sea otter that was cleaned with detergents remained high (Costa and Kooyman 1981). One of the two otters returned to 16 °C holding tanks eventually died, while both otters returned to 20 °C holding tanks recovered.

6.4.1.9 Indirect Effects

There is no available information regarding the indirect effects of oil spills on either sea otters or polar bears. However, Stirling et al. (1978) suggested that because ringed and bearded seals are the primary prey of polar bears, major declines in seal abundance as a result of an oil spill would probably be reflected in changes in the distribution and possibly the abundance of polar bears.

6.4.2 Vulnerability

Marine-associated mammals could be exposed to oil present in shoreline habitats, near ice edges in the Arctic, and in marine or estuarine waters adjacent to these habitats. The vulnerability of otter populations would probably be highest during periods when individuals are congregated in shoreline areas and both air and water temperatures are low. On the other hand, polar bears would probably be most vulnerable to oil spills during periods when they are found near ice edges or open leads, since oil may accumulate in these areas and their primary prey (ringed and bearded seals) may be present in relatively large numbers. This combination of circumstances would increase the potential for contamination of swimming bears, as well as the potential for oil ingestion through the consumption of contaminated prey.

SECTION 6.5

SHOREBIRDS AND WADING BIRDS

Wayne Duval

Birds have often been most visibly affected by oil spills, although the degree of contamination and mortality has been highly dependent on the circumstances surrounding the spills (e.g. the time and location of spills in relation to seasonal distribution patterns and associated habitat use by bird populations) (Duval et al. 1981). Case histories of spills have also indicated substantial variability in the sensitivity and vulnerability of different species (Holmes and Cronshaw 1977; Duval et al. 1981). The most serious effects of past spills have been on seabirds (particularly alcids), while shorebirds and wading birds have been affected less frequently and in smaller numbers.

Available literature regarding the fate and effects of oil on seabirds is summarized in Section 2.2. The behavioural, physical, physiological and pathological effects of oil exposure, as well as available data on developmental and reproductive implications of oil exposure are also examined there. Much of the research in these areas may be relevant to shorebirds and wading birds but is not repeated here. This section reviews the limited number of oil spills where contamination or mortality of shorebirds or wading birds was observed by on-site research teams. However, it should be emphasized that the types and species of birds affected by oil spills have not always been identified in the literature, and still other literature has focused on those bird populations affected in greatest numbers. Consequently, shorebirds and wading birds may have been affected by more spills than those identified below.

6.5.1 Sensitivity to Oil

Follow-up studies were conducted by Ramseier et al. (1973) after a gasoline and diesel fuel storage tank rupture and spill at Deception Bay (Hudson Strait) in Northern Quebec during June 1970. Of a total of eight dead birds, seven were shorebirds (plovers and sandpipers) that had been extensively contaminated with oil. The fact that oil released from the ruptured tanks contaminated backshore and shoreline areas before reaching the bay may have contributed to the extensive contamination of these birds, since contamination of shorebirds would be less severe if they were only feeding on a shoreline where oil was stranded as a result of a marine spill. An unspecified number of oil-contaminated herons were also observed during studies initiated after a spill of diesel oil and Bunker C from the tanker WITWATER in February 1969 (Rutzler and Sterrer 1970). Pelicans and plovers were also affected by crude oil released following the grounding of the OCEAN EAGLE in March 1968 (Center for Short-Lived Phenomena 1975). Both of the latter spills occurred in tropical waters rather than in the cold marine waters that are the focus of this review, and neither group of investigators provided information on the actual number of birds affected, or whether these individuals died or survived after contact with oil.

A more rigorous examination of the effects of oil on shorebirds and wading birds was completed following a spill of Bunker C from a storage tank near the Hackensack River estuary in New Jersey during May 1976. The research team involved in the damage assessment reported that both wading birds and shorebirds were affected by the bunker fuel (Mattson et al. 1977), although no estimates of mortality (if any) were provided. Wading birds contaminated by oil were black-crowned herons (Nycticorax nycticorax), least bittern (Ixobrychus exilis), snow egret (Leucophoyx thula) and the Florida gallinule (Gallinula chloropus). An abandoned gallinule nest was also found to contain oiled eggs. Since the vegetation surrounding this nest was free of oil, Mattson et al. (1977) concluded that oil was transferred to the eggs from the breast feathers of the parent birds. Most shorebird species were unaffected by this spill because they did not feed in habitats that were contaminated with oil. However, the yellow-legged sandpiper (*Totanus melanoleucus*) apparently contacted free-floating oil due to its habit of walking in breast-deep water to feed on killifish. Mattson et al. (1977) reported that none of the oiled sandpipers observed were incapable of strong flight, but suggested that some of the oiled birds may have been unlikely to complete their northern migration to tundra breeding grounds.

6.5.2 Vulnerability and Recovery

In the Canadian Arctic, nesting shorebirds in coastal areas would be most vulnerable to oil spills during August and September when the adults or juveniles of some species stage in the littoral zone (ESL 1982b). Since phalaropes are habitual swimmers, they would also be vulnerable to spills during foraging activities and migration. ESL (1982b) suggested that phalaropes, sanderlings and ruddy turnstones would probably require the longest recovery period following population decreases associated with oil spills because these species are considered more vulnerable to littoral zone disturbances than other shorebird species (Connors et al. 1979).

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